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HEALTH
OF THE
NORTHWEST ATLANTIC

A Report to the Interdepartmental Committee on Environmental Issues

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INTRODUCTION

R.C.H. Wilson*

The earliest fossil remains of living organisms are associated with ancient seas. From the time of his oldest written records, man has sailed in ships over the sea and exploited the resources beneath its surface, building a modern society with cultural and economic characteristics strongly influenced by the sea and its biological foundation. Today, despite every school-child's knowledge that oceans occupy about two-thirds of the world's surface, despite the fact that about 6.5% of the animal protein consumed in Canada[1] comes from the sea, concern for the health of the ocean is still not widespread. The sea has tended to preoccupy only those who made a living by it. Even for its most intimate acquaintances, its obvious size and power have allayed most fears that man's activities will change it in a meaningful way.

Since confederation, the ocean and its users have been under the study or control of federal government departments housed in the provinces bordering the Atlantic coast. The regional heads of these departments together comprise the Interdepartmental Committee on Environmental Issues. Some records, lobster catch statistics for example, have been maintained since the last century. Other information, such as environmental levels of organic contaminants, is much more recent in comparison. The foci of study and control expand and shift to reflect the pressures of new patterns of technological development, resource exploitation and transportation.

The western North Atlantic has traditionally been used as a fishing grounds. In 1977 the fisheries of the four Atlantic Provinces plus Quebec together totalled one million tonnes and about two-thirds of the marketed value of all Canadian fish[2]. While expectations of future catch include increases for some species, the overall yield is not expected to change much in the period 1980-85. The fastest growing sector of the industry is probably aquaculture, where the number of operations has risen from 16 in 1975 to 59 in 1983. The commercial fishery is dependent on a satisfactory level of environmental quality. Abnormally high levels of pollutants would not only threaten the viability of fish stocks but could also lead to accumulations which would prevent the sale of fish or fish products for human consumption. The culture of shellfish is especially vulnerable in this respect, in that it is a coastal activity whose water quality requirements conflict with the common use of estuaries and bays to treat domestic sewage.

Development of the oil and gas resources off the Atlantic Canadian coast has been an important regional issue for at least five years. Exploration activity in recent years has remained fairly steady, with 15 wells drilled in 1982, but it is expected to double by 1985. Exploration is occuring

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mainly on the Scotian Shelf, the Grand Banks off Labrador, and in the Gulf of St. Lawrence. Development drilling is imminent for the Venture gas structure off Sable Island and the Hibernia oil structure on the Grand Banks. Environmental risks center on the possibility of an accident during drilling, production or transportation. While studies of oil spills elsewhere generally indicate that long term impact is highly localized and can be almost eliminated by thorough cleanup, chronic impacts from spills and from normal operational inputs have been documented.

While the offshore oil and gas industry is perhaps the most visible threat to ocean quality, it is not the most important. A compilation of statistics on the input of pollutants showed that atmospheric and riverine inputs were the largest sources of nutrients and toxic metals to the North Sea[3], despite the intensity of the offshore oil and gas industry. Similar statistics for the western North Atlantic are unavailable, but it is to be expected that the major inputs in this region would be associated with industries located onshore.

The papers comprising this volume were assembled to bring together information about some of the pressures on organisms off the Canadian Atlantic coast. The levels and distribution of chemical contaminants, especially some distance from shore, are of concern to scientists and regulators because trends in this area are measured and influenced more slowly than in terrestrial ecosystems. The oceans, too, are part of the global commons to whose management and preservation national governments are generally committed. Knowledge of contamination and the response of marine communities to it is an essential part of the Canadian strategy for their protection.

The focus of the papers which follow thus tends towards the offshore rather than the estuaries and coastal embayments where the better known and perhaps more serious pollution problems exist. By necessity arising from their scope, the papers are all fairly general but they indicate, I believe, that scientists working in the Atlantic Region have not detected any serious disturbances to the health of the Northwest Atlantic. This does not mean that government and industry can sit back and relax; the need for discovery, information, discussion and agreement will continue to challenge institutional abilities if the present state of quality is to be maintained.

The papers which follow are of three types. The initial four papers set the stage by generally describing the regional oceanography, and the geological, atmospheric and terrestrial processes which influence marine water quality. The middle section contains a substance-by-substance presentation of the sources, levels, distribution and effects of physical and chemical contaminants. Nutrients are notably missing from this list; the reason is that eutrophication is a problem nowhere in the Atlantic offshore[4]. The final paper describes the response by government and industry to the perceived threat of ocean contamination.

It is obvious from the foregoing that the papers presented here do not represent the last or the most complete word on any particular subject. Scientists interested in important gaps in existing knowledge and specialists looking for a thorough exposition are generally aware of where to find these facts. This compilation should fill a need for those seeking a general overview of regional marine environmental quality in the Canadian Atlantic offshore.

Conclusions from the study of marine pollution have been obtained at considerable cost. Variability in the levels of natural or introduced substances has dictated difficult sampling methodologies and lengthy baseline periods to detect trends. A simple change in analytical procedure has frequently caused an older data set, obtained with scrupulous, painstaking care, to be discarded in the light of more modern understanding. Authors who attempt to summarize a specialist's knowledge over an area as large as the Northwest Atlantic must deal with technical constraints such as these and must also face more general problems. Even in the length of time that it takes to prepare and review a collection of papers such as this, information that was correct and up-to-date when written sometimes becomes dated. Also, as in any field of science, interpreters of marine quality data must apply judgement to reach their conclusions, and their opinions and values may not be universally shared. For these and other reasons, the opinions expressed by the authors in this compendium are their own, and do not necessarily represent the official views of the departments for which they work.

REFERENCES

1. Willmer, J. 1983. Personal communication. DFO, Ottawa.

2. Scarratt, D.J. (ed.). 1982. Canadian Atlantic offshore fishery atlas. Can. Spec. Publ. Fish Aquat. Sci., 47 (Revised), 121 p.

3. Norton, R.L. 1982. Assessment of pollution loads to the North Sea. Tech. Rep. TR 182, Water Research Center, Medmenham, U.K., 28 p.

4. Coote, A.R. 1981. Personal communication. Atlantic Oceanographic Laboratory, DFO.

OCEANOGRAPHIC SETTING

Lawrence P. Hildebrand *

INTRODUCTION

Canada has one of the longest coastlines in the world, with a total mainland length of about 30,000 km. Approximately one-third of the coastline lies along the Atlantic seaboard, extending from the Gulf of Maine at the Canada-U.S. border to the northern tip of Labrador (Latitude 42° N to Latitude 60° N) (Figure 1). Along this coastline stretches the Continental Shelf, which is about 200 km wide off Nova Scotia, reaches out 500 km on the Grand Banks southeast of Newfoundland, and then narrows to less than 100 km along the coast of Labrador. Included in this region are the Scotian Shelf, Grand Banks, Newfoundland and Labrador Banks and the inland waters of the Gulf of St. Lawrence and the Bay of Fundy. In general, the water depths are between 75 and 250 m, with the exception of the Laurentian Channel (450 m), which passes from the mouth of the St. Lawrence River to the edge of the Continental Shelf, and some shallower areas less than 50 m on Georges Bank, Sable Island Bank and on the south tip of the Grand Banks.

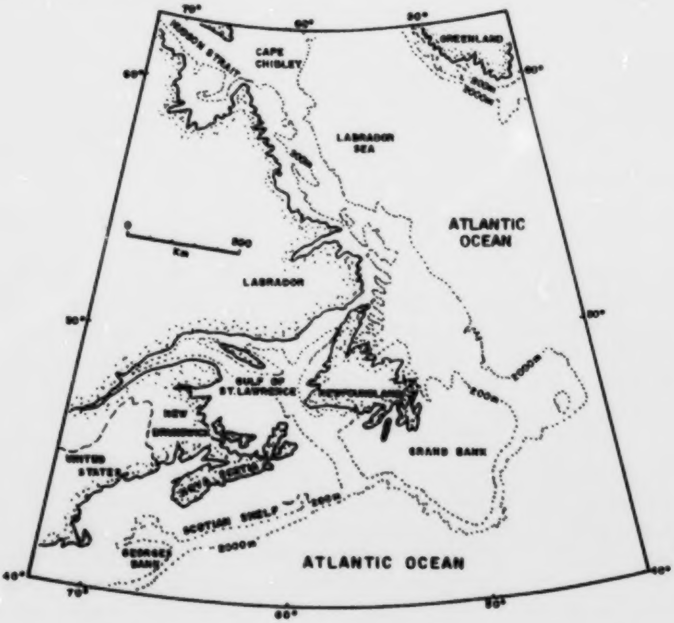
70°
60°
50°
HUDSON STRAIT
CAPE CHIDLEY
GREENLAND
60°
60°
200m
2000m
LABRADOR SEA
200m
ATLANTIC OCEAN
0
500
Km
LABRADOR
50°
50°
GULF OF ST.LAWRENCE
NEWFOUNDLAND
NEW BRUNSWICK
2000m
200m
UNITED STATES
NOVA SCOTIA
GRAND BANK
SCOTIAN SHELF
200m
GEORGES BANK
2000m
ATLANTIC OCEAN
40°
40°
70°
60°
50°

Figure 1: *The Canadian Atlantic coast and continental shelf* [1].

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The Continental Shelf north of Cape Cod is more than usually dissected by channels and basins, due principally to its recent glacial history. The Gulf of Maine and Scotian Shelf are characterized by a series of isolated banks and deep basins, which are, respectively, above and below the global average for Continental Shelf depths. To the north, the whole of the large flat area of the Grand Banks, with the exception of the coastal Avalon Channel, is shallower than most Continental Shelf regions of comparable size, mostly lying between 50 and 100 m and having an unusually pronounced platform shape. Further north still, the Continental Shelf extending the length of the coast of Labrador is highly atypical in being almost twice the depth, on average, of a typical Continential Shelf.[2]

PHYSICAL OCEANOGRAPHY

Currents

Along with the Oyashio/Kuroshio Region off the north coast of Japan, the Northwest Atlantic may be one of the most highly variable parts of the world oceans. This is a result of the apposition of the warm pole-ward Gulf Stream, the cold equator-ward Labrador current and the important role of the immense and variable spring discharge from major rivers and from melting ice in the low Arctic[2].

The major surface currents of the Atlantic east coast and adjacent deep ocean are indicated in Figure 2. It shows the tendency of the water to move southward over the Continental Shelf areas, and the presence of nearly closed circulation patterns called gyres on Flemish Cap, in the Gulf of St. Lawrence and on Georges Bank. The process of exchange between coastal and deep ocean waters on the Scotian Shelf and at the tail of the Grand Banks, and the proximity of energetic oceanic currents such as the Gulf Stream to the Continental Shelf, is also evident. It appears that the Continental Shelf areas off Nova Scotia, Newfoundland and Labrador and the Gulf of St. Lawrence are all interconnected by these currents[17].

To give an impression of the magnitude of these flows, the volume transport of the major currents is given in Table 1, as well as the time it would take for a flow of those strengths to fill an empty Gulf of St. Lawrence.

TABLE 1 - MEAN CURRENTS OFF THE ATLANTIC EAST COAST

Current	Transport ($10^6 m^3/s$)	Time to Fill Gulf
River runoff in Gulf of St. Lawrence	0.02	58 years
Nova Scotian	0.35	3 years
Labrador	5.0	3 months
Gulf Stream	50	1 week

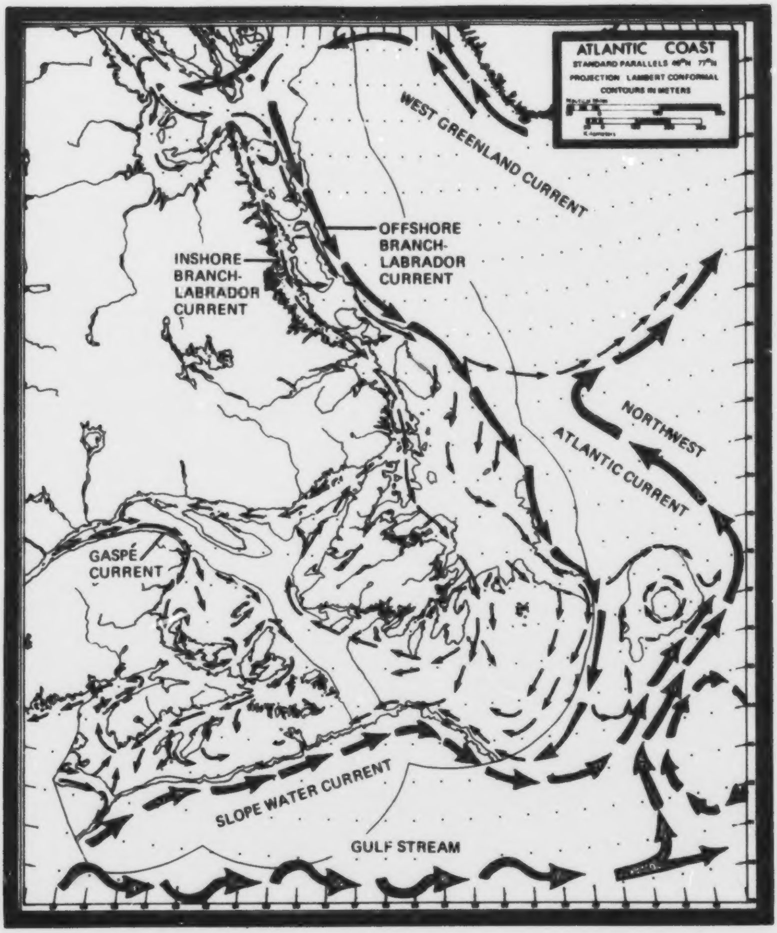

ATLANTIC COAST
STANDARD PARALLELS
PROJECTION LAMBERT CONFORMAL
CONTOURS IN METERS
WEST GREENLAND CURRENT
OFFSHORE BRANCH-LABRADOR CURRENT
INSHORE BRANCH-LABRADOR CURRENT
NORTHWEST ATLANTIC CURRENT
GASPÉ CURRENT
SLOPE WATER CURRENT
GULF STREAM

Figure 2: *General surface circulation: cold currents, solid arrows; warm currents, stippled arrows. Arrow length approximates current velocity: long arrows, 50-100 cm / s; medium arrows, 30-50 cm / s; short arrows, less than 30 cm / s.*[3].

The Labrador current begins at the southern end of Davis Strait where cold Arctic waters are joined by the somewhat warmer, saltier waters of the west Greenland current which moves across Davis Strait from the western coast of Greenland. Part of this current enters Hudson Strait where it mixes with the outflow from Hudson Bay and exits along the south side of the Strait, where it rejoins the main mass of water moving southward along the Labrador coast. The main current consists of an inshore branch which is largely composed of waters originating from Hudson Bay and the Arctic, and a stronger offshore branch composed largely of waters from the west Greenland current. While these two components can be distinguished, there is significant exchange between them.

Off Newfoundland, the Labrador current flows southward onto the northeast Grand Banks and around and over the Newfoundland Grand Banks. The main current flows along the western side of Flemish Pass and southward to the tail of the Grand Banks. Much of the water turns offshore in meanders and eddies, eventually mixing with the Northwest Atlantic current and flowing northward again. An inshore branch flows through the Avalon Channel and then turns westward at the edge of the Continental Shelf where it is joined with waters from the Grand Bank. A comprehensive bibliography of the physical oceanography of the Labrador region has been prepared by Dobson and Jordan[4].

Within the Gulf of St. Lawrence, surface water exits from the estuary region as a strong coastal current (the Gaspé current) which flows eastward along the Gaspé coast, around and over the Magdalen shallows and exits the Gulf of St. Lawrence on the western side of Cabot Strait. There is a concurrent inflow in the deeper layers on the Newfoundland side. The fresh water runoff which is a major driving force of the Gulf circulation varies seasonally with a high in May and a low in January and February. The general circulation in the Gulf of St. Lawrence has been treated extensively by El-Sabh[5].

Waters exiting from the Gulf flow westward along the coast of Nova Scotia within 20 km of shore (Nova Scotia current) and eventually enter the Bay of Fundy and Gulf of Maine. Offshore, the flows are generally weaker and more variable in direction, with a considerable offshore component. The Nova Scotia current is modified as it moves southward, increasing in salinity, heat content and nutrients as a result of exchanges with offshore waters.

Offshore, the warm slope water current flows eastward along the Continental Shelf south of Nova Scotia and Newfoundland to the tail of the Grand Banks. Further south, the Gulf Stream also flows to the east, splitting at the tail of the Bank. One branch turns northward and joins with the slope water current to form the Northwest Atlantic current, which flows northward past Flemish Cap and loops into the southern Labrador Sea before flowing eastward across the North Atlantic.

While the Gulf Stream is, on average, several hundred kilometers south of the Continental Rise, it actually does not flow due east, but proceeds eastward as a series of large (order 100 km in diameter) continually changing loops or rings. Gulf Stream rings are episodic, there being an average 6 per year which break off to form warm core eddies which may impinge on the Continental Rise, causing an exchange between coastal and offshore waters.

The circulation, variability and dynamics of the Scotian Shelf and Slope have been reviewed by Smith, Petrie and Mann[6] and the general circulation of the Gulf of Maine - Bay of Fundy areas have been discussed by Bumpus[7], and Brown and Beardsley[8].

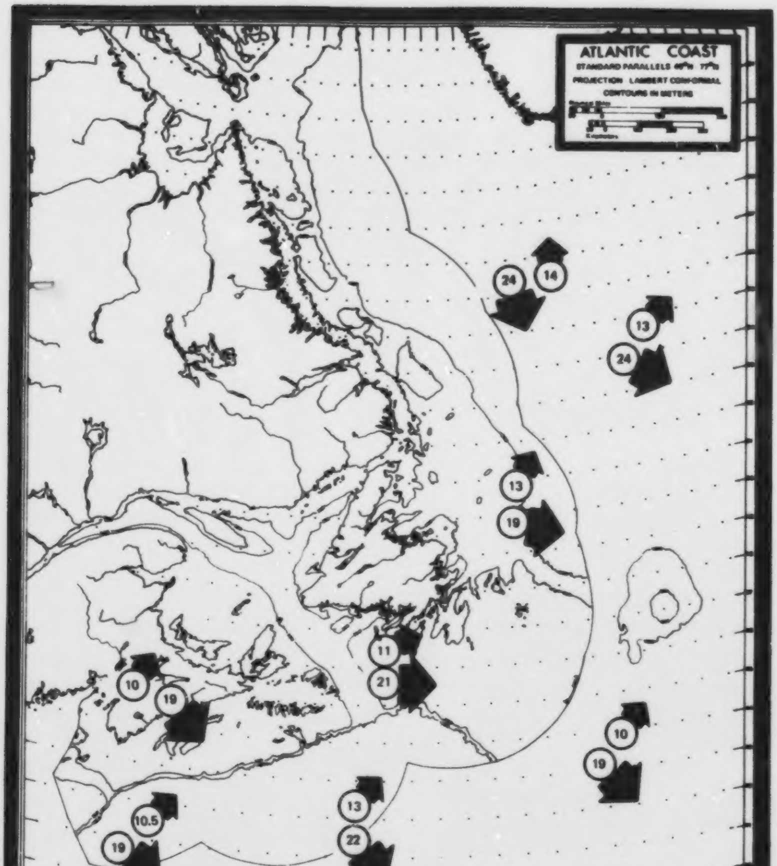
Winds and Waves

The Canadian east coast weather and physical oceanography form an intricately related system. The marine climate of the area is dominated by the movement of low pressure zones across the area. In winter, there are three main storm tracks: A north one moving out from the Great Lakes basin, another from Cape Hatteras to Cape Cod coastal plain, and a third one from the Florida area. These storm tracks bring lows in succession, normally about six to ten a month, and in their wake they produce gale to storm force winds. A number of these storms slow down as they reach this area, and their influence can extend for a week or more over large areas of the eastern Canadian offshore. During the summer months these storms move north with decreased numbers and intensity. The accompanying pressure patterns generally favour mean winds from the northwest and west in the winter months, with mean speeds ranging from 18 to 24 knots. Summer winds over the Canadian Atlantic come from a southerly direction and are generally more moderate, with mean speeds ranging from 10 to 14 knots[3].

Clearly these wind conditions have a great effect upon wave generation in both the coastal regions and the offshore. The sea state varies greatly with season and location. Consequently, during the winter months (January and December) the wave energy may be 5 to 6 times greater than during the summer, and over the Grand Banks, 3 to 4 times that over the Scotian Shelf.

The reason for this lies in the fact that, during the winter, strong winds from the northwest are either directed offshore with regard to the southern coast, thereby reducing the sea state by opposing the mid-Atlantic waves, or are parallel to the eastern seaboard where they generate large seas along the coast of Labrador and over the Grand Banks. During the summer, winds are primarily from the southwest along the southern coast and away from the eastern seaboard. Since these winds are light, wave action along both coastlines is relatively low.

Fully exposed to wave action from the south are Georges Bank at the entrance to the Bay of Fundy, the Scotian Shelf, and the southern portion


ATLANTIC COAST
STANDARD PARALLELS
PROJECTION LAMBERT CONFORMAL
CONTOURS IN METERS
Kilometers
24
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10.5
19

Figure 3: ***Prevailing winter and summer winds. Winter (January) winds, large arrows; summer (July) winds, small arrows; monthly mean wind speeds (knots), encircled*** [3].

of the shelf off Newfoundland. The Grand Banks, southeast of Newfoundland, are open to nearly every direction, with the exception of a sector from the northwest, while the east coasts of Newfoundland and Labrador are open to waves from the Labrador Sea and the North Atlantic.

As the depths on the Continental Shelf are usually greater than 75 m, it can be assumed that waves move freely over most of the Continental Shelf without being noticeably affected by depth. This, however, does not apply to some shallower areas on Georges Bank, around Sable Island, and on the south tip of the Grand Banks where depths are less than 50 m and where waves experience considerable shoaling and refraction. This is also true in the nearshore regions where shoaling affects almost the entire wave spectrum. For a complete review of wave activity in this region see Neu[9].

Fog

Canada's east coast is noted for its fog. The fog is caused by warm air passing over the cold water and is most common during the summer. The highest incidence occurs along the Atlantic coast from the Bay of Fundy to the Avalon Peninsula. The incidence falls off sharply to the north and west with the result that most of the Gulf of St. Lawrence has less than one-half the number of foggy periods experienced by the Atlantic coastal margin.

Temperature and Salinity

The temperature of sea water varies from place to place and from season to season in the eastern Canadian offshore. Less obvious are changes in the salinity of the water in the open sea where land runoff exerts less influence. These variations depend primarily on precipitation and the amount of evaporation, but also on vertical mixing, as both temperature and salinity vary with depth, location and time.

In describing temperature-salinity relationships, eastern Canadian waters have been divided into three zones. The Low Arctic zone includes areas influenced by the west Greenland and Labrador currents. August temperatures are about 4°-10°C, and salinities are in the range of 31-34°/oo, with values as low as 29°/oo seasonally. The Boreal zone includes the areas strongly influenced by the North Atlantic Current, such as the Scotian Shelf and eastern Grand Banks. August surface temperatures are between 10°-19°C, with salinities of 31-35°/oo (less on the Scotian Shelf and in the southern Gulf of St. Lawrence) overlapping with those of the low Arctic zone. The Cool Subtropical zone includes the waters of the North Atlantic Current. August surface temperatures are about 19°-23°C, and salinities are over 35°/oo.

The boundaries of these zones are not exact and fluctuate seasonally: the southern Gulf of St. Lawrence, for example, changes from an ice-covered area in winter to one with surface waters of 16°C or more in summer. The water in the Gulf of St. Lawrence is strongly influenced by the fresh water discharged by the St. Lawrence River, which is the major river flowing into this area. Water of low salinity flows out of the Gulf and southwest along the coast of Nova Scotia where it mixes with modified North Atlantic Current water. The waters of the Scotian Shelf are thus markedly less saline than those of the Grand Banks further east.

In the Gulf of St. Lawrence, there are three main layers of water in the summer. There is a warm layer in the deep channels with temperatures in the range of 2°-5°C, and salinities of about 35°/oo, sufficient to make this the heaviest layer despite the warm temperatures. Overlying this deep layer, at depths of 50 to over 100 m, is an intermediate cold layer with temperatures of 0°C or lower, and salinities of 30-33°/oo. The thickness of this layer is at a maximum in spring and a minimum in the fall because of varied mixing with the surface layer. In the surface layer, temperatures are dependent upon the season. During the spring and summer, temperatures warm progressively to as high as 20°C at the surface but remain at 0° to 5°C at depths of 20-30 m.

Waters of the Labrador current, or those derived from it, occupy the remainder of the Canadian coastal area. This is a simple cold flow with salinities in the 30-35°/oo range. Temperatures in the surface layer vary from -1.6° to -1.0°C in winter in all areas to 10° - 15°C in Newfoundland waters in summer and 3° - 8°C along the Labrador coast. In the subsurface layers, temperatures remain less than 0°C year round. This cold current is confined to the Continental Shelf, and beyond the slope warmer and more saline water is present. In general, the coldest and least saline water lies near the surface and close to shore.

Tides

In the Atlantic Region of Canada there is a very wide spectrum of tidal range[10,11]. In the far northern areas the tidal range is in the order of 9 to 12 m, but throughout the rest of the area it is generally in the order of a meter or so, with the exception of the Bay of Fundy tides of up to 12 to 16 m, among the highest tides in the world. The smallest tides in the region occur in the southwestern part of the Gulf of St. Lawrence where the average range is less than 1 meter and spring tide seldom exceeds 1.8 meters. Slightly larger ranges occur in the eastern half of the Northumberland Strait, the Bay of Chaleur and in the approaches to the Strait of Belle Isle. By comparison, the North Sea has tides ranging from 2 to 3 meters.

Ice

A large part of Canada's east coast experiences the seasonal effects of ice[12]. Ice begins to form to the north of Labrador in November/December and moves slowly south at about 10 miles per day, reaching the Grand Banks in February/March, often as far south as Latitude 46°N. The natural thickness is in the order of a meter, but wave action creates ridges and hummocks which often generate thicknesses of 3 to 5 meters. These can persist for quite some time, even after they reach the Grand Banks. Southern progression is maximized normally around April/May, when the

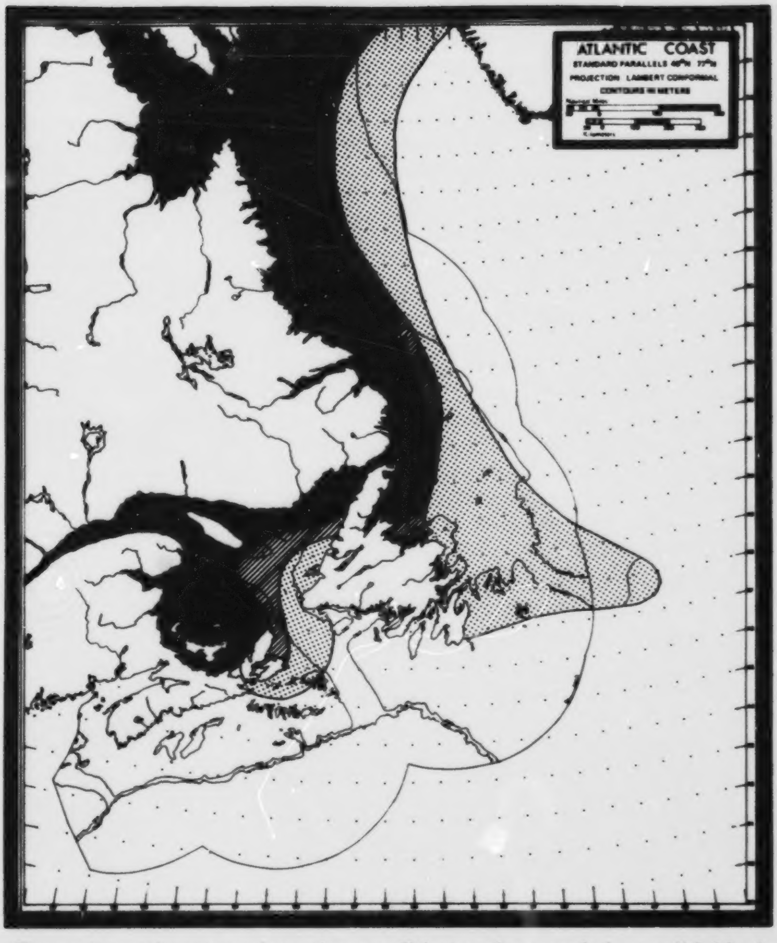

ATLANTIC COAST
STANDARD PARALLELS
PROJECTION LAMBERT CONFORMAL
CONTOURS IN METERS

Figure 4: *Average winter ice conditions: close pack ice, solid black; open pack ice, diagonal; maximum extent of all ice, stippled*[3].

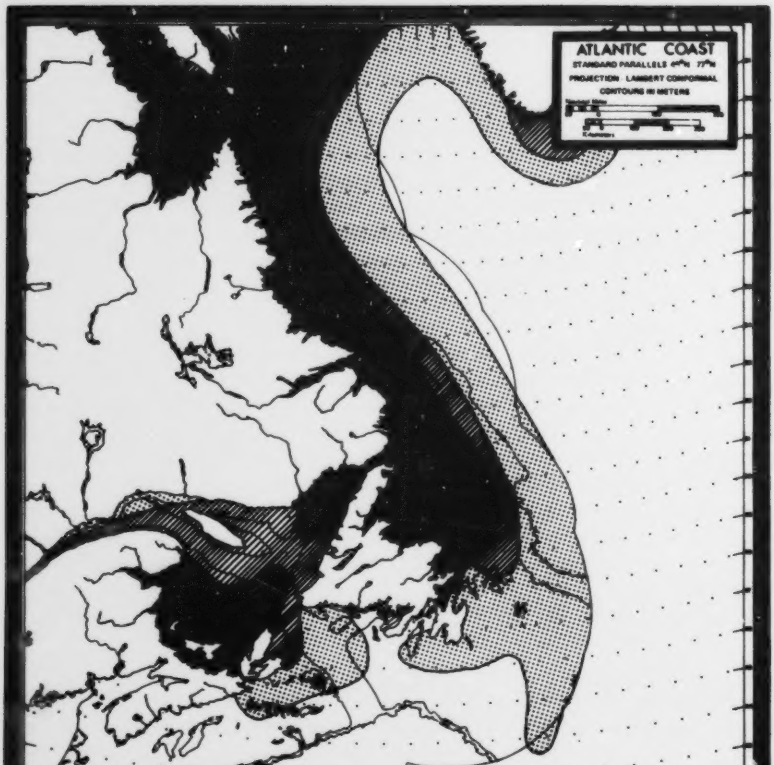

ATLANTIC COAST
STANDARD PARALLELS 77°N
PROJECTION LAMBERT CONFORMAL
CONTOURS IN METERS
Kilometers

Figure 5: *Average spring ice conditions: close pack ice, solid black; open pack ice, diagonal; maximum extent of all ice, stippled*[3].

winds tend to become westerly and the ice is driven eastward into warmer waters and melts, starting a northward retreat of the pack. It clears the Belle Isle Strait about the end of May in most years and the northern Labrador coast about the third or fourth week in July.

Within the rest of the region, essentially ice-free conditions are found in the Bay of Fundy (except for the upper reaches) and along the mainland coast of Nova Scotia. Seasonal ice of local origin is found in the Gulf of St. Lawrence and Cabot Strait, beginning in December and reaching maximum extent in late February. The ice stream departs the Gulf and spreads south and southwest toward Sable Island, frequently completely covering the banks north of the Island for short periods. When the fields have their greatest extension, the ice spreads east-northeastward toward the southern shore of Newfoundland, though seldom actually reaching the coast. Eastern and northern Newfoundland and the Labrador coastal waters are affected by local ice in the early part of the winter, but ice floes from the northern waters form an increasing proportion of the pack during late winter and spring.

Icebergs are another prominent feature of this region. They are produced continuously in the north, and over the past 50 years an average of 380 bergs per year have drifted south of the 48° parallel. Approximately 85% of these icebergs are generated from the west coast of Greenland, another 10% from the east coast, and the remaining 5% from the Canadian Arctic Archipelago. Iceberg densities vary considerably from north to south and with season, depending on the water temperature, sea state and pack ice extent. They range in size from a few thousand tons to about 10 million tons, but only about ten a year of the largest bergs on average reach the Grand Banks. When they come to the Grand Banks, the dominant flow is around the edge of the banks rather than across, and of the few that do arrive, most move around the few hundred meter contour.

BIOLOGICAL OCEANOGRAPHY

Primary and Secondary Production

As the Canadian east coast physical oceanographic system is all interrelated, so is the biology. Much of what happens in the east coast biological system is in response to the spring bloom of the phytoplankton. The timing and size of this bloom is controlled on regional and local scales by the interactions between light and nutrient availability, which are in turn the result of physical oceanographic and meteorological processes.

The classical mechanism to account for the onset of phytoplankton blooms in the spring and their subsequent evolution depends on the balance between the intensity of solar incident energy, thermal stratification and the photosynthetic and respiratory rates of plant cells. However, this in itself is not sufficient to explain the exceptionally high productivity which characterizes all continental shelves, including those of eastern Canada.

On the continental shelves, where tidal currents are sufficiently powerful, a tidally mixed region comes to be separated from a thermally stratified region by a tidally-induced front within which the thermocline of the stratified region intercepts the surface. As a result, the region of the tidal front itself commonly has a higher and more sustained rate of production because nutrients are continually supplied from below the thermocline into the surface zone of high illumination.

In the Northwest Atlantic, tidally-induced fronts are known to exist in the Gulf of Maine, delimiting tidally mixed regions over Nantucket Shoals and Georges Bank, around southern Nova Scotia and Grand Manan Island, and in the Bay of Fundy above a line approximately between Saint John and Digby. In the Gulf of St. Lawrence, fronts of high chlorophyll production occur in the Jacques Cartier Passage, in the Strait of Belle Isle, in the St. Lawrence estuary, the Northumberland Strait and around the Magdalen Islands.

The relative strength of the phytoplankton bloom in the spring increases progressively northwards along the Continental Shelf, so that it is a much more prominent feature off Newfoundland than in the Gulf of Maine. On the Scotian Shelf and Grand Banks there is a distinct second bloom in autumn which is not seen in the Gulf of Maine. A similar but separate trend is seen offshore so that the seasonal signal in phytoplankton biomass is strongest in the Labrador Sea and weakest in the slope water region south of the Grand Banks. The spring increase begins around the first of March in all areas except the Labrador Sea region, where it is about 30 days later, and in the open ocean south of the Grand Banks, where the spring bloom occurs very early.

The Nova Scotia shelf-break front is unusual in that biological activity generally increases seawards across the Continental Shelf.

The Grand Banks shelf-break front appears to induce relatively high concentrations of chlorophyll throughout the period of March to November due to the presence to the east of a captured clockwise gyral current centered above Flemish Cap. Associated with the thermal front around this gyre is an area of nutrient enrichment and high biological productivity. A similar situation exists on Georges Bank in the Gulf of Maine.

The general region of the shelf-break fronts is also associated with aggregations of biota higher in the food chain. This is evidenced by a concentration of plankton-eating whales, sea birds and demersal fish.

In the Northwest Atlantic we recognize four principal kinds of plankton distributions. Warm water oceanic species are most common in the Gulf Stream and in the North Atlantic Drift water, whereas cold water oceanic species occur right across the North Atlantic, north of the oceanic polar front, in the western half of the ocean. The two remaining groups are the continental shelf species and the coastal species which inhabit embayments and estuaries.

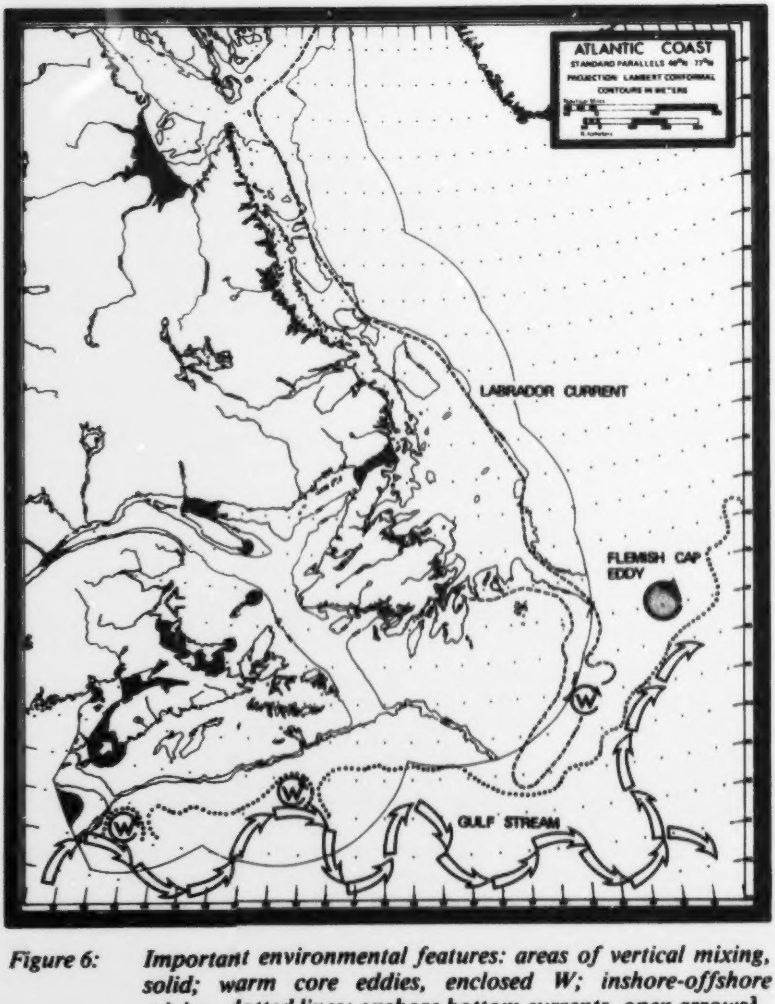

ATLANTIC COAST
STANDARD PARALLELS
PROJECTION: LAMBERT CONFORMAL
CONTOURS IN METERS
LABRADOR CURRENT
FLEMISH CAP EDDY
W
W
W
W
GULF STREAM

Figure 6: ***Important environmental features: areas of vertical mixing, solid; warm core eddies, enclosed W; inshore-offshore mixing, dotted lines; onshore bottom currents, open arrows***[3].

It is important to note the apparently universal fact that while there are several hundreds of species (even of a single group like copepods) present in a single plankton ecosystem, only a very few dominate numerically and by biomass. In one collection of oceanic plankton containing 106 species of copepods, it was found that only a dozen species made up 98.9% of the biomass. Commonly only two or three species comprise more than 90% of the biomass in a macroplankton ecosystem.

Preliminary analysis of sequential collections of plankton from the same sites clearly indicates sustained changes in the plankton ecosytem, closely resembling those now well described in the eastern Atlantic. The trends show the striking nature of the between-year variability, reflected in the fact that the spring bloom had progressively advanced in this region by nearly one month during the 1960's. The evidence suggests that the changes observed in the plankton ecosystem over these continental shelves are consequences of events occurring on the scale of the whole ocean basin of the North Atlantic.

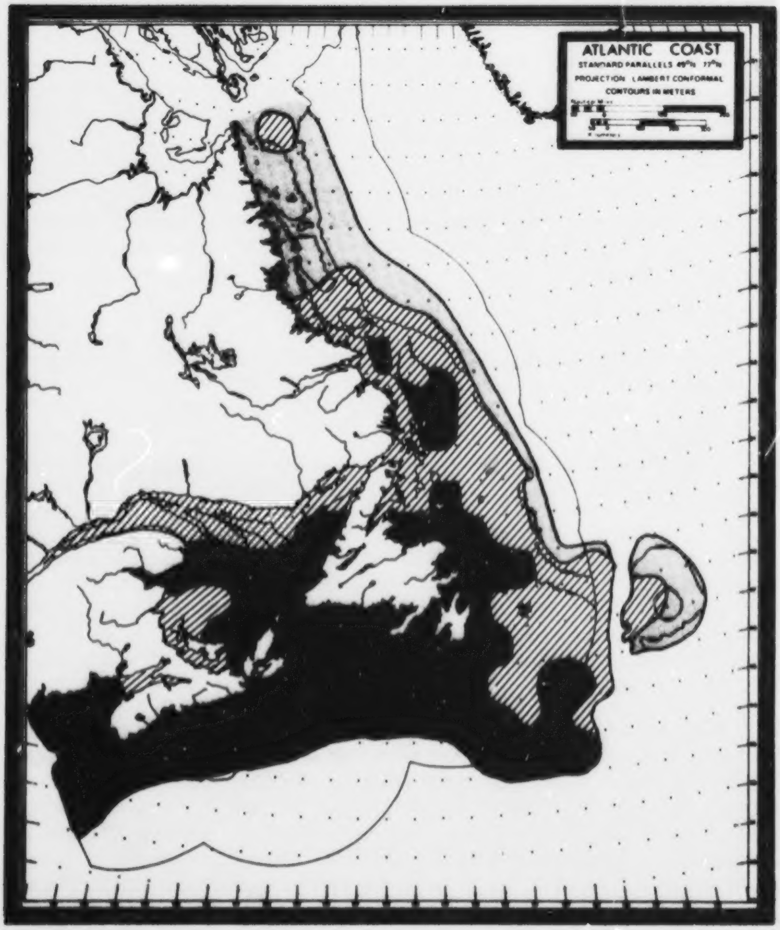
Fish Resources

Variability has been a dominant characteristic of the physical environment of this region and is closely related to the patterns of variability which appear so clearly in the primary and secondary production processes. Thus, it is hardly surprising that extreme variability is also of fundamental importance in our attempts to understand the distribution and abundance of the fisheries resource.

Of particular importance in this respect are depth, temperature and bottom type. Each fish species has a range of preferred depths and temperatures. A difference of water temperature of as little as 1°C can dramatically affect the distribution of some temperate fish species, especially where the change occurs near the upper or lower limits of their range. Likewise, the distribution of some groundfish and shellfish can be related to bottom type[18].

The ecology of larval and juvenile fishes in such temperate marine environments as the Canadian Atlantic is usually totally different to that of the adults. In the case of the younger stages and particularly the eggs and larvae, the early life history is more oriented toward life near the surface than to the bottom. Their biology is intricately tied to the physical, chemical and biological oceanography of the system in which they live. Thus, though we may have a general understanding of where and when spawning takes place (and this varies quite considerably from year to year), the subsequent movements of the eggs and/or larvae are dependent on the variable winds and currents, and the rate of development (and thus length of sojourn in the near surface waters), which is in turn dependent on the water temperature and the percentage survival under natural conditions.

Figure 7 indicates that, in terms of the present commercial fishery, areas of high resource importance include the area from the mouth of the



ATLANTIC COAST
STANDARD PARALLELS 49°N 77°N
PROJECTION LAMBERT CONFORMAL
CONTOURS IN METERS

Figure 7: *Key resource areas: high importance, solid; medium importance, diagonal; minor importance, stippling*[3].

Bay of Fundy, across Georges Bank, and the Scotian Shelf. This area is extended to include the southern Gulf of St. Lawrence, the eastern shore of Newfoundland, and the area south of Newfoundland from inshore to the edge of the continental shelf, and from St. Pierre Bank to the southwestern edge of the Grand Bank. Also important is the Hamilton Inlet Bank off Labrador. The fisheries of Newfoundland, Nova Scotia, New Brunswick and Quebec together account for about two-thirds of the marketed value of all Canadian fish.

Groundfish make up more than one-half of the catch in the eastern Canadian offshore. Total allowable catches (TACs) for 1981 for the traditional major groundfish species (cod, redfish, haddock, pollock, American plaice, witch, Greenland halibut and yellowtail) in this region amounted to 884,000 tonnes, the Canadian share of which was 775,000 tonnes.

Among the main Canadian pelagic species (hake, herring, mackerel, bluefin tuna, capelin, swordfish, Atlantic salmon), the overall finfish TAC is projected to increase between 1981 and 1987 from 1.50 to 2.04 million tonnes.

The decision by the Government of Canada to extend fisheries jurisdiction to 200 miles was given emphasis by a fisheries resource crisis off the Canadian Atlantic coast. Severe resource declines were a major factor which contributed in 1974 to threaten the survival of the Canadian Atlantic groundfish industry. A major objective of the 200 mile management regime is to rebuild the resource through stringent conservation measures so as to provide increased catches and catch rates for Canadian fishermen. Since 1977, most of the major fish stocks within the Canadian Atlantic zone have been managed to permit stock rebuilding.

A complete description of specific stocks and their projections is contained in two publications of the Department of Fisheries and Oceans: "Resource Prospects for Canada's Atlantic Fisheries, 1981-1987"[13] and the "Canadian Atlantic Offshore Fishery Atlas[3]".

MARINE MAMMALS

The numbers and biomass of marine mammals off the Canadian east coast both exceed the human population in the Atlantic Provinces and the eastern Arctic[19]. The populations of individual species are spread over a wide geographic range, which is extended in many species as a result of seasonal migrations. These animals come to the east coast because it is a productive feeding ground.

Twenty-two species of whales occur off Canada's Atlantic coast, as well as six species of seals.[14] Several species (harbour seal, ringed seal, harbour porpoise and minke whale) are coastal species found along the eastern seaboard. These occur year round and breed in the Newfoundland-northern Grand Banks area. For the harp seal and hooded seal, the ice edge

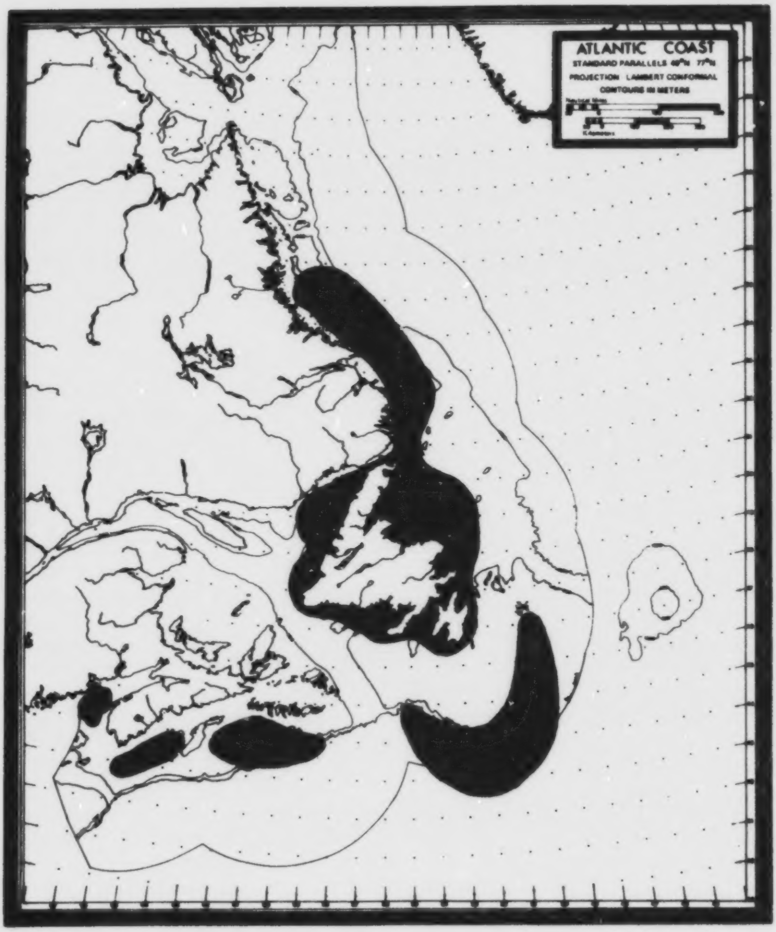

ATLANTIC COAST
STANDARD PARALLELS
PROJECTION LAMBERT CONFORMAL
CONTOURS IN METERS
Nautical Miles
Kilometers

Figure 8: *Cetaceans: key areas, solid black*[3].

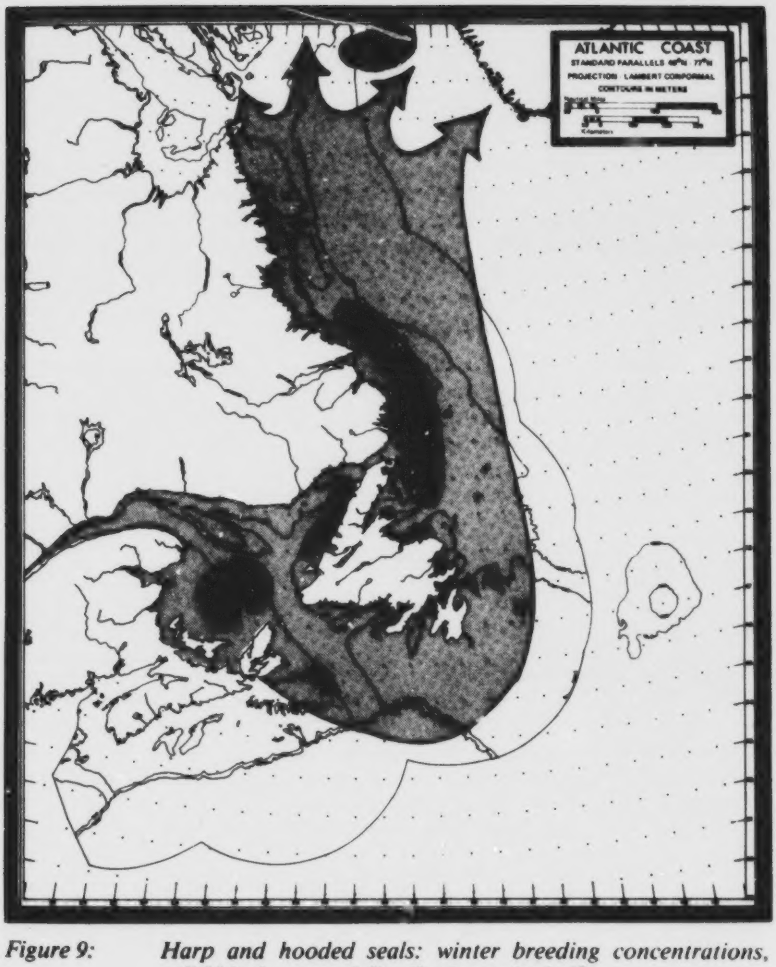

ATLANTIC COAST
STANDARD PARALLELS
PROJECTION · LAMBERT CONFORMAL
CONTOURS IN METERS

Figure 9: *Harp and hooded seals: winter breeding concentrations, solid black; general distribution, stippled*[3].

in the Gulf of St. Lawrence and the "front" (north of the Strait of Belle Isle) are the principal breeding areas of the species in Canadian waters.

Other marine mammals which may be found in the area are seasonal transients. The pilot whale, sei whale and fin whale are found frequently in Newfoundland waters during the summer, and the major part of the fin whale population summers here. Grey seals disperse from their Nova Scotia and New Brunswick breeding areas to feed in summer around Newfoundland. Blue whales, sperm whales and killer whales are seen in the area only as they move northward to their summer ranges, and the northern bottlenose whale may occur in the area at the southern limit of its winter range.

The distribution of most species is related to food, though for several species, most notably the ringed seal, the availability of suitable habitat appears to be important. The harp seal and hooded seal require ice-edge habitat and migrations of the young seals result from their association with the retreating ice edge. The major whelping ground of the hooded seal occurs in the area. Nearly all whelping of the harp seals occurs in the first two weeks of March, slightly earlier in the Gulf than in the Front.

Most mammal species in the area are opportunistic feeders. Pelagic species eat mainly fish and squid and nearshore ones include benthic invertebrates in addition to fish. Often food preferences exist. The primary food of nearshore pilot whales is squid and an onshore pattern of movement is observed with the coastal movement of squid. Baleen whales feed on plankton and small fish.

A more complete description of the eastern Canadian marine mammals can be found in Leatherwood *et. al.*[15]

MARINE BIRDS

The sea bird populations which breed from northern Labrador south to Nova Scotia, including all the ones which breed in the four Atlantic Provinces and on the Quebec shore of the Gulf of St. Lawrence, are considered to be part of the same geographic unit. The view becomes even broader outside the breeding season, since the area is visited by non-breeding sea birds from as far away as Hudson Strait, Greenland, Russia and Antarctica[20].

There is no clear definition of the word "sea bird", but it is generally taken to include such groups as the petrels, gannets, cormorants, phalaropes, gulls, terns, jaegers and auks, but to exclude groups such as loons, grebes and ducks. Twenty-one species of sea bird breed in this region, the commonest of which are leach's storm-petrels, northern gannets, double-crested cormorants, great black-backed and herring gulls, black-legged kittiwakes, common and Arctic terns, razorbills, common and thick-billed murres, black guillemots and Atlantic puffins.

By far the most important area is the island of Newfoundland, where there are at least three million breeding sea birds, mainly along the east and south coasts between Funk Island and Cape St. Marys. Over 300,000 birds breed in Labrador, mainly in the region of Groswater Bay but with a small group of colonies near Nain. Some 200,000 birds breed on the Quebec shores of the Gulf of St. Lawrence. By contrast, the numbers in Nova Scotia, Prince Edward Island and New Brunswick are much smaller, yielding a combined total of little more than 10,000 birds (excluding the herring gulls and great black-backed gulls).

Several factors have contributed to this rather lop-sided distribution; geology is one of them. Most colonial sea birds prefer to nest on steep, rocky islands where they are relatively safe from predators; however, this habitat type is virtually non-existent in the three Maritime Provinces. Oceanographic factors are also important. Newfoundland, Labrador and the Quebec north shore are, to a greater or lesser degree, influenced by the Labrador Current, where capelin (an important food fish for most sea birds) is most abundant. Man has directly and indirectly altered these distribution patterns over the last two hundred years.

The six largest colonies in the region are located at Funk Island, Cape St. Marys, and the Witless Bay Islands in Newfoundland, the Gannet Islands off southeast Labrador, Bonaventure Island off Gaspé, and Bird Rocks in the Gulf of St. Lawrence. In a broader perspective, the number of birds and the variety of species seem about average for a region on the borderline between the low Arctic and boreal-temperate zones. The sea bird population is richer than the sea bird community on the Murmansk Coast (Barents Sea) and the low Arctic community in Hudson Strait, but poorer than that of Britain and Ireland. Nevertheless, all of the gannets and most of the razorbills and puffins in North America, and most of the eastern population of common murres are found along the Atlantic Canadian coast.

Brown and Nettleship[16] examined these eastern Canadian marine bird populations and noted that these populations fluctuate in size. The gannet population on Bonaventure, for example, initially increased after the colony was protected some 60 years ago but has now begun to decline again, probably from a combination of human disturbance and infertility caused by the accumulation of pesticide residues. Murres and puffins seem to be declining throughout North America, probably from a combination of factors including oil pollution, to which these birds are especially vulnerable, drowning in fishing nets, legal and illegal over-hunting and competition with fishermen for species such as capelin.

Kittiwakes on the other hand are increasing, as are herring and great black-backed gulls. Fulmars colonized Newfoundland for the first time at the beginning of the 1970's, as did Manx shearwaters a little later, and the black-headed gull almost certainly breeds there as well.

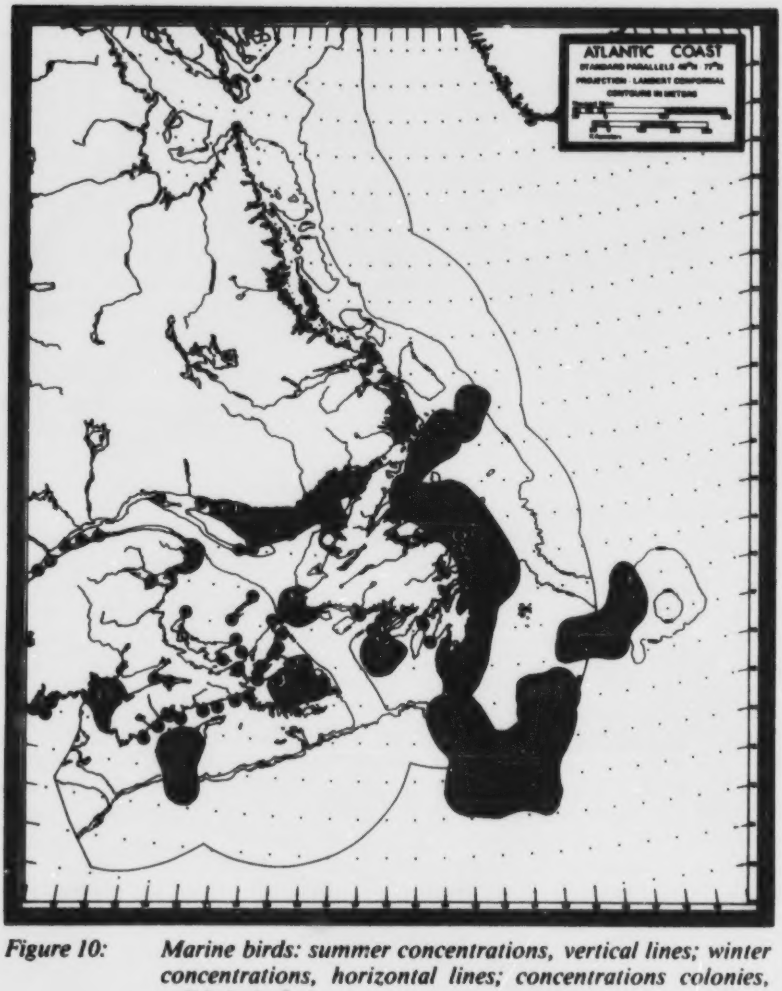

ATLANTIC COAST
STANDARD PARALLELS
PROJECTION - LAMBERT CONFORMAL
CONTOURS IN METERS

Figure 10: *Marine birds: summer concentrations, vertical lines; winter concentrations, horizontal lines; concentrations colonies, solid circles* [3].

None of these estimates takes into account the very large population of subadult birds which are present during the summer. Sea birds are long-lived animals under natural conditions and most species do not begin to breed until they are four years old or more.

The number of sea birds off Atlantic Canada increases enormously outside the breeding season. The most important wintering area is off eastern Newfoundland, where the sea is relatively ice-free, and where there is a highly productive plankton bloom which begins early in the year. There is also a population of capelin there in winter as in summer. Smaller numbers of birds winter on the Scotian Shelf, and probably off Labrador before the ice forms early in the new year. There is little activity in the Gulf of St. Lawrence, even while it is still ice free.

SUMMARY

Although this chapter has presented only a brief introduction to a range of topics which are far more extensive than discussed here, it has hopefully served to introduce the reader to the concept of the extensive variability and the intricate relationships between and within the physical and biological components of the eastern Canadian offshore. The impression one should be left with is that the balance within this system is a delicate one and any effects which might alter even a single component could have serious repercussions in other areas, leading to more far-reaching consequences which may not now be fully comprehended.

ACKNOWLEDGEMENT

This paper has relied heavily on contributions to the "Workshop on Environmental Considerations of East Coast Offshore Hydrocarbon Development", held in St. John's, Newfoundland in December, 1980. Extensive use has been made of material prepared for the workshop proceedings by Longhurst[2], Petrie[17], Sandeman[18], Brodie[19], Brown[20] and Allen[21]. Their contribution is gratefully acknowledged.

REFERENCES

1. Neu, H. J.A. 1972. Extreme wave height distribution along the Canadian Atlantic coast. Ocean Industry, 7 (7): 45-49.

2. Longhurst, A.R., 1980. Biological oceanography. In: Offshore Environment in the 80's; A Workshop on Environmental Considerations of East Coest Offshore Hydrocarbon Development. Proceedings, 1980. St. John's, Newfoundland, Dec. 2-4, 1980.

3. Scarratt, D. J. (ed.). 1982. Canadian atlantic offshore fishery atlas. Can. Spec. Publ. Fish Aquat. Sci., 47 (revised): 101 pp.

4. Dobson, D. and F. Jordan. 1978. The Labrador Sea: A bibliography of physical oceanography. Bedford Institute of Oceanography, Data Series, BI-D-78-5, 25 pp.

5. El-Sabh, M. I. 1975. Transport and currents in the Gulf of St. Lawrence. Bedford Institute of Oceanography Report Series, BI-R-75-9, 180 pp.

6. Smith, P.C., B. Petrie and C. R. Mann. 1978. Circulation, variability and dynamics of the Scotian shelf and slope. J. Fish. Res. Board Can., 35 (8): 1067-1083.

7. Bumpus, D. 1976. A review of the physical oceanography of Georges Bank. ICNAF Res. Bull., 12: 199-234.

8. Brown, W.S. and R.C. Beardsley. 1978. Winter circulation in the western Gulf of Maine, Part 1: Cooling and water mass formation. J. Phys. Oceanogr., 8:265-277.

9. Neu, H.J.A. 1976. Wave climate of the North Atlantic, 1970. Bedford Institute of Oceangraphy Report series, BI-R-76-10.

10. Canada Department of Fisheries and Oceans. (Annual). Canadian Tide and Current Tables, Volume I, Atlantic Coast and Bay of Fundy. Ottawa.

11. Canada Department of Fisheries and Oceans. (Annual). Canadian Tide and Current Tables, Volume 2, Gulf of St. Lawrence. Ottawa.

12. Markham, W. E. 1980. Ice Atlas-Eastern Canadian Seaboard. AES Publ., En-56-55: 95 pp.

13. Canada Department of Fisheries and Oceans. 1981. Resource Prospects for Canada's Atlantic Fisheries, 1981-1987. Ottawa.

14. Sergeant, D. E. 1961. Whales and dolphins of the Canadian East Coast. Can. Fish. Res. Board, Can. Arct. Unit, Circ. 7:17 pp.

15. Leatherwood, A., D. K. Caldwell and H. E. Winn. 1976. Whales, dolphins and porpoises of the western North Atlantic; A guide to their identification. Natl. Ocean, Atmos. Adm. Tech. Rep., National Marine Fish Service Circular, 396: 172 pp.

16. Brown, R .G. B. and D. N. Nettleship. 1982. Capelin and seabirds in the Northwest Atlantic. An investigation associated with the program "Studies on Northern Seabirds". Seabird Research Unit, Canadian Wildlife Service, Environment Canada (Report No. 127).

17. Petrie, B., 1980. Physical characteristics of the marine ecosystem: An Atlantic east coast example. In: Offshore Environment in the 80's; A Workshop on Environmental Considerations of East Coast Offshore Hydrocarbon Development. Proceedings, 1980. St. John's, Newfoundland, Dec. 2-4, 1980.

18. Sandeman, E.J., 1980. The fish and shellfish resources of the offshore area. In: Offshore Environment in the 80's; A Workshop on Environmental Considerations of East Coast Offshore Hydrocarbon Development. Proceedings, 1980. St. John's, Newfoundland, Dec. 2-4, 1980.

19. Brodie, P.F., 1980. Marine mammals in the ecosystem of the Canadian east coast. In: Offshore Environment in the 80's; A Workshop on Environmental Considerations of East Coast Offshore Hydrocarbon Development. Proceedings, 1980. St. John's, Newfoundland, Dec. 2-4, 1980.

20. Brown, R.G.B., 1980. Marine birds. In: Offshore Environment in the 80's; A Workshop on Environmental Considerations of East Coast Offshore Hydrocarbon Development. Proceedings, 1980. St. John's, Newfoundland, Dec. 2-4, 1980.

21. Allen, J., 1980. The physical environment in which offshore development will take place. In: Offshore Environment in the 80's; A Workshop on Environmental Considerations of East Coast Offshore Hydrocarbon Development. Proceedings, 1980. St. John's, Newfoundland, Dec. 2-4, 1980.

GEOLOGICAL PROCESSES AND FACTORS

David J. W. Piper*

GEOLOGICAL SETTING

Bedrock

The inner part of the continental shelf off eastern Canada is underlain by rocks that are a continuation of those seen on the adjacent land, while further offshore is a sedimentary sequence many kilometers thick that has accumulated in the 100-180 million years since the opening of the Atlantic Ocean and Labrador Sea.

Old hard folded rocks of Appalachian orogen outcrop on the inner part of the Scotian Shelf off the Atlantic Coast of Nova Scotia[1], and off parts of southern and western Newfoundland (notably between the Avalon Peninsula and Virgin Rocks). The inner part of the Labrador Shelf is underlain by a continuation of the resistant Precambrian rocks on the Canadian Shield.

Flat-lying sedimentary rocks of lower Paleozoic age underlie the northern part of the Gulf of St. Lawrence including the Strait of Belle Isle, and the southeast Baffin Shelf. Sedimentary rocks of Carboniferous age accumulated in a series of basins, the margins of which are seen onshore, and contain the gypsum, salt and coal of Atlantic Canada. The largest basin underlies all but the northern third of the Gulf of St. Lawrence; the Sydney Basin lies between Cape Breton Island and southern Newfoundland; and a third major basin lies off the northeast coast of Newfoundland.

The Bay of Fundy and adjacent parts of the Gulf of Maine are underlain by rocks of late Triassic age, accumulated in a rift basin immediately prior to the opening of the central north Atlantic Ocean in early Jurassic times. With the development of the Atlantic Ocean, the Mesozoic-Cenozoic sediment wedge of the outer part of the Scotian Shelf and southwestern Grand Banks began to form. The oldest rocks include salt, which now forms salt domes beneath the continental slope. This is overlain by predominantly carbonate Jurassic rocks, and sandstones and shales of Cretaceous age deposited in a large delta system associated with an ancestral St. Lawrence River. Tertiary strata are principally shales. Off Labrador, the Mesozoic-Cenozoic wedge is younger, being principally of Cretaceous and Tertiary age. East of Newfoundland, Jurassic strata were gently tilted and eroded prior to desposition of the younger Cretaceous and Tertiary sequence.

Most outer shelf areas are underlain by Tertiary strata. These thin landwards, and on many central shelf areas Cretaceous rocks reach close to the surface and are overlain by thin glacial deposits. Jurassic strata are only rarely exposed.

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Most large scale geomorphologic features can be related to the gross geology described above. Land areas of Atlantic Canada are largely built of old hard Pre-Cambrian and lower Palaeozoic Applachian rocks. The major embayments of the Bay of Fundy and Gulf of St. Lawrence correspond to Triassic or Carboniferous sedimentary basins. The line of outer banks on the shelf are similar to escarpments developed on land in bedded Mesozoic-Cenozoic strata onshore on the eastern seaboard of the U.S.A., and were probably formed under conditions of fluvial erosion in the later Tertiary.[2,3]

Glacial and Post-Glacial Sediments

Most, if not all, of the continental shelf off eastern Canada has been covered with glacial ice at some time during the last 2 million years. This glaciation has had three major effects on the offshore area.

(1) Many of the most prominent morphological features of the continental shelf are the result of glacial erosion, which took place principally during occasional major glacial advances.

(2) Much of the surface of the continental shelf is blanketed by glacial till and outwash from the last (Wisconsinan) glaciation, between 10,000 and 70,000 years ago, and sedimentation on the continental slope and in deeper water has been profoundly influenced by glacial events on the continental shelf.

(3) Changes in sea level of over 100 m associated with glaciation uncovered shallower areas of the shelf and led to profoundly different oceanographic conditions.

The effects of glacial erosion are most clearly seen in the excavation of deep troughs and channels on the continental shelf. The Labrador Marginal Trough, which parallels the Labrador coast and reaches depths in excess of 900 m, marks the boundary of the soft Mesozoic-Cenozoic sediments with the hard Pre-Cambrian bedrock of the Canadian Shield: glacial erosion appears to have stripped off the soft rocks above the unconformity. Although this erosion may in part be fluvial, the overdeepening must be the result of glaciation.

The major transverse channels of the continental shelf, such as the Northeast Channel and the Laurentian Channel at either end of the Scotian Shelf, and Hawke, Cartwright, and Hopedale Saddles on the Labrador Shelf have characteristic straight trends, and steep walls of glacial valleys and are cut one to several hundred meters below the level of the outer shelf banks. Smaller shelf channels and basins (LaHave and Emerald Basins, Hermitage and St. Pierre Channels for example) probably also derive from glacial erosion. Erosion of all these major geomorphologic features appears to predate the last (Wisconsinan) glaciation, which was less extensive than some earlier glaciations.

The coastal fjords of Newfoundland and Labrador have been overdeepened by glacial erosion: the major Newfoundland fjords floored with soft Carboniferous sediments have depths of up to 500 m. On a smaller scale, coastal inlets on the Nova Scotian coast, such as Bedford Basin and St. Margaret's Bay, show the effects of glacial overdeepening. The very large amount of small scale morphological irregularity on the inner parts of the Scotian, northeast Newfoundland and Labrador Shelves, developed on old hard bedrock, is principally the result of glacial erosion excavating irregular areas of less resistant rock. This irregular marine topography, with innumerable islands and shoals, is a continuation of the very irregular land topography which is highlighted by the abundance of lakes. As on land, a much smoother topography is developed on the softer Carboniferous, Triassic, and Cretaceous-Tertiary bedrock areas.

Many areas of the continental shelf are floored by deposits of glacial till: a poorly sorted, only partly consolidated deposit of debris transported by glacial ice and deposited either at the bed of the grounded glacier, or by melting out of a floating ice shelf or icebergs. Most of the till seen on the continental shelf dates from the last Wisconsinan glaciation and is a few tens of meters in thickness, but older till probably occurs in some other shelf areas. Till has been generally removed by erosion in water depths of less than 100 meters.

The extent of glacial ice on the continental shelf during the Wisconsinan glaciation is uncertain. Ice may have covered much of the Scotian

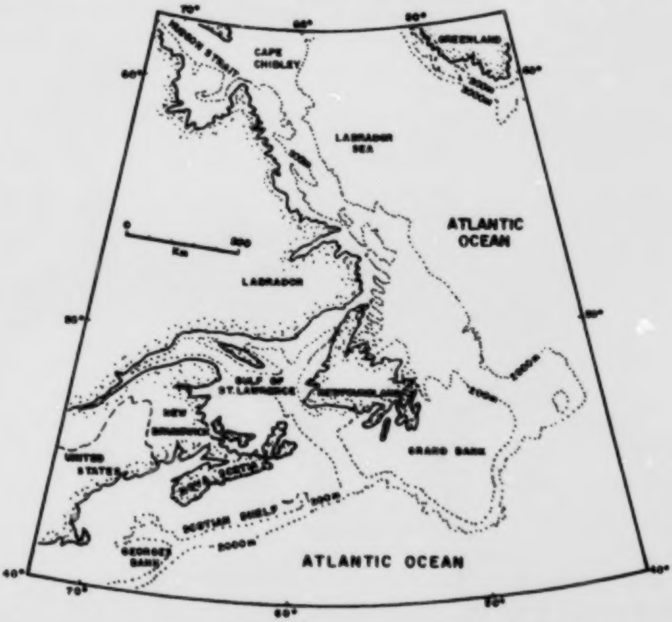
70°
60°
50°
HUDSON STRAIT
CAPE CHIDLEY
GREENLAND
LABRADOR SEA
ATLANTIC OCEAN
0
200
Km
LABRADOR
GULF OF ST. LAWRENCE
NEWFOUNDLAND
NEW BRUNSWICK
UNITED STATES
NOVA SCOTIA
GRAND BANK
SCOTIAN SHELF
1000m
2000m
GEORGES BANK
ATLANTIC OCEAN
40°

Figure 1: *The Canadian Atlantic coast and continental shelf.*

Shelf some 40,000 years ago, but by 14,000 years ago it had retreated to approximately the present coastline. Wisconsinan ice was probably less extensive on much of the Labrador Shelf, and had retreated to the present coastline by 11,000 years ago.

A complex sequence of unconsolidated sands and muds, of variable thicknesses up to many tens of meters, rests on glacial till. The origin of this sequence is best known on the Scotian Shelf [3,4], which is here used as an example, but analagous deposits occur on other continental shelf areas off eastern Canada. All of the formations described below outcrop at the sea bed at one place or another off eastern Canada.

The Emerald Silt directly overlies till. It consists of finely bedded mud with local silt, sand and scattered pebbles, and coarsens towards moraines of till, with which it interfingers. It was deposited under brackish to marine conditions, at least in part beneath floating ice.[3]

In shelf basins, the LaHave Clay overlies Emerald Silt. It is a homogeneous dark silty clay that is up to 70 m thick. It has accumulated in marine conditions following deglaciation in the last 14,000 years.

The Sambro Sand occurs as a peripheral deposit around the Scotian Shelf banks in water depths exceeding 115 m. It consists of moderately sorted sand or sandy gravel up to 100 m thick, apparently reworked from the till and Emerald Silt. In shallower water, the Sable Island Sand and Gravel is a very well sorted sand veneer less than 20 m thick that overlies eroded remnants of till and Emerald Silt.

The contrast in sorting between the Sambro Sand and the Sable Island Sand reflects the role played by changing relative sea levels. During glaciation, relative sea level fluctuates as a result of the removal of water from the world ocean to form ice on land, and the load effects of this ice on the earth's crust [5]. On the outer part of the continental shelf, sea level attained a maximum lowering of 115 m some 15,000-18,000 years ago and has been steadily rising since that time. The Sambro Sand accumulated below this sea level, while the Sable Island Sand and Gravel developed its distinctive character in the surf zone and shallow water as sea level rose.

Closer to the areas covered with glacial ice (such as the northern Gulf of St. Lawrence and the coast of Labrador), the load effects of the former ice predominate and relative sea level has fallen continuously since deglaciation as the crust has recovered from the load.

Surficial Distribution of Sediments

As a result of this complex geological history, the distribution of sea bed types is also complex. Bedrock outcrops over much of the innermost shelf and coastal zone, together with late glacial sediments (Emerald Silt equivalents) in areas of falling relative sea level. Sand accumulates in beaches and in shallow coastal bays flushed by waves or tidal currents (the latter affecting Minas Basin and Northumberland Strait) while mud occurs in deeper coastal basins and on tidal flats on sheltered coasts.

Mid-shelf basins are floored with contemporary muds (LaHave Clay equivalents), but often have late glacial sediments (Emerald Silt equivalents) outcropping at their margins. Till outcrops are common on the mid and outer shelf, yielding a hard cobbly bottom often with relict iceberg scours. Shallow outer shelf banks have a veneer of well sorted sand and gravel. (Sable Island Sand and Gravel equivalents), while deeper banks have less well sorted sand (Sambro Sand equivalent). Sand extends to water depths of several hundred meters on the continental slope, and then grades into mud in deeper water.

SEABED EROSION AND TRANSPORT OF SEDIMENT

Introduction

If we examine the seabed, clear geologic evidence for erosion is widespread. What is less clear is the age of this erosion, since relict erosional features may be preserved with little modification as sea level rises. Many erosional events (such as major storms or iceberg scars) are rare, and therefore difficult to observe.

Waves and currents are the most widespread erosional agents: they are considered separately for three different environments — the wave dominated nearshore zone, the tide dominated nearshore zone, and the offshore. Bioturbational resuspension of sediment is equally ubiquitous. Scour by icebergs and slumping of steep slopes are erosional events that are restricted both in time and space.

Waves and Currents

Erosion by waves and currents in nearshore environments is a well recognized phenomenon. Entire beaches may be destroyed by storms. Tidal channels and sand bars shift position. Techniques such as precise resurvey and depth of disturbance rods[6] show that, nearshore, sand in water depths of less than 20 m is frequently remobilized to depths of tens of centimeters. Beaches may be eroded to depths of meters during storms, thereby releasing buried pollutants such as infiltrated oil[7].

The character and amount of erosion in offshore areas is very poorly known. In the shallow offshore area of Sable Island Bank there is circumstantial evidence for the migration of large sand megaripples (dune-like sand bodies), with associated reworking of sediment to depths of tens of centimeters or meters, but there is at present no quantitative understanding of the process.

Side-scan sonographs and bottom photographs show that areas of sandy seabed to water depths in excess of 100 m are partly covered by sedimentary structures such as ripple marks that suggest sediment mobility. At Hibernia, ripple marks in 100 m of water change during storms[8]. Larger bedforms occur in places, but they are believed to be relicts from times of lowered sea level, and are now probably immobile[3] although direct evidence for this is lacking. On the outermost continental shelf and upper continental slope[9], current scour marks are found around boulders in water depths as great as 350 m. Evidence of this type suggests that in areas of sandy sea floor in water depths of more than a few tens of meters, sediment mobility is probably restricted to the upper few centimeters, taking the form of winnowing of areas of positive relief and the migration of ripples.

It is possible that occasional large storms may remobilize much greater amounts of sediment. Measured current velocities in winter in water depths of 150 m frequently exceed 1 m/sec on the Scotian Shelf[10]; sustained currents of this magnitude in shallow water can produce bedforms that rework to depths of over a meter. In southern Emerald Basin, Kontopoulos and Piper[11] found evidence for occasional episodic sand-mud depositional events with a recurrence interval of around 200 years, suggesting that the erosional effects of occasional extreme storms may be hard to document because of the time periods involved. Piper *et al.*[12] inferred a depth of reworking of 1.5 m in a sand core from 44 m water depth off Liverpool, Nova Scotia. The preservation of relict late glacial bedforms on the shelf[3] suggests, however, that if major sediment remobilization does occur, it is probably not ubiquitous, but restricted to certain areas of the continental shelf.

There is even less evidence for whether muddy areas of the seabed suffer periodic erosion and resuspension. Deep water basins such as Emerald Basin are generally assumed to be quiet depositional environments which act as sediment traps without any re-erosion by currents. In shallower water, sediment distribution patterns in the Bay of Fundy change through time[13,14] suggesting sediment movement in water depths of tens of meters. In mud-floored nearshore basins in Mahone Bay, Letson[15] found a 20 cm thick surface layer of soft mud at water depths of 20 m, which appeared to be remobilized during storms with a 10-50 year recurrence interval. In Bedford Basin and Halifax Harbour, there is circumstantial evidence that muddy basin margin slopes are reworked in water depths of 50 m, although the depth of reworking is probably only a few centimeters[16].

The picture that emerges from these scattered data is that a surface layer of sediment, perhaps up to 10 cm thick, is relatively mobile on the continental shelf, particularly in areas of clean sandy seabed and on steeper slopes and basin margins. Remobilization to much greater depths may occur in water depths of less than 50 m, and areas of current constriction and intensification, but is not of widespread occurrence. Locally elevated areas, whether natural or man made, are particularly susceptible to scouring and reworking.

Bioturbation

Organisms burrow in and rework surficial sediments. There have been few systematic studies of bioturbation off eastern Canada: the notable exceptions are the work of Risk and Yeo[17] in the tidal flats of the Bay of Fundy and of Risk *et al.*[18] in Chedabucto Bay. However, results of studies in other areas can be extrapolated with considerable confidence.

The depth and intensity of reworking by burrowing organisms depends on the species present and their abundance. In sandy bottoms, the depth of burrowing (particularly in shallow water) appears related to the depth of reworking by waves and currents. In water depths of more than 10 m, bioturbation frequently reworks the upper 3-10 cm; while in shallow subtidal environments[17], there are abundant burrows to depths of 15 cm.

Similar variations are found in muddy environments, but with generally higher rates of reworking and occasional very deep burrows. Risk *et al.*[18] found open shrimp burrows to sub-bottom depths of 3 m in muddy bottom in a water depth of 5 to 20 m in Chedabucto Bay, and calculated that complete reworking of sediment would take place in 70 years. Rates of reworking of the surface ten centimeters, several orders of magnitude higher, have been found in shallow subtidal and estuarine muds, where bioturbation increases the water content of muds making them more susceptible to erosion by weak tidal currents. Physical resuspension by both filter feeding organisms and the movement of large animals such as crustaceans and fish may play a large role in gradual movement of muddy sediment on slopes[9].

Iceberg Scour

Iceberg scour is a potent means of seabed reworking in areas influenced by the Labrador Current with water depths of less than 200 m[19]. Grounded icebergs plough up sediment in grooves and form craters where they come to rest, and this disturbed sediment may then be susceptible to reworking by waves, currents and organisms. For example, in the Hibernia area[8] scours over the last 10,000 years have disturbed the seabed to a sub-bottom depth of at least 3 m, but most scours are less than 0.5 m deep. On average, only about 1% of the seabed has been disturbed by iceberg scours in the last 10,000 years. In areas such as the Labrador Shelf banks and the Strait of Belle Isle, contemporary iceberg scour reworking is more intense.

Slumping

Slumping or mass movement of sediment occurs most frequently on steep seabed slopes. Such steep slopes occur at the edges of coastal fjords, at the margins of some glacially excavated basins and channels, and on the continental slope. Evidence from sidescan sonographs and high resolution seismic profiles for slumping on the continental shelf is rare, but possible small examples are known from the Labrador Shelf[20].

Slumping is extremely widespread on the continental slope, but most slump features appear many thousand years old and related to glacial conditions. The only well-documented slump in historical times was associated with the magnitude 7 earthquake of 1929 at the southeast margin of the Laurentian Channel [21]. Slumping occurred over an area of 2×10^4 km^2 of continental slope, removing surface sediment from perhaps 60% of that area, yielding a total sediment volume of 10^9-10^{10} m^3. There was presumably at the same time some slumping on equally steep slopes on the continental shelf within 100 km of the epicentre, but this has not yet been demonstrated by field observation. There is some evidence for smaller scale slumping within the last few thousand years on parts of the Scotian and Labrador slopes.

SEDIMENT DEPOSITION ON THE CONTINENTAL SHELF

Introduction

Areas of contemporary sediment deposition are potential sinks for environmental pollutants. Knowledge of sediment sources and budgets may provide information on the transport paths of pollutants, and the rates at which they may be removed from the environment. As with erosion, it is useful to distinguish nearshore from offshore areas. In addition, we must recognize that the ultimate sink for much of the sediment transported on the continental shelf is the deeper water continental slope and rise.

Sediment Sources

Unlike most well studied areas of the world, rivers are unimportant sources of sediment to the sea floor off eastern Canada. Even large rivers like the St. Lawrence and Churchill have exceptionally low sediment loads, and river derived sediment is largely trapped within coastal estuaries. Coastal erosion of glacial till[22], soft bedrock[23] or late glacial marine deposits in areas of crustal rebound following glaciation[24] are the principal sources of sediment to the coastal zone. Mud and, to a lesser extent, sand can escape from the coastal zone to the inner shelf, principally by tidal currents, but the magnitude of this effect is uncertain, even in the well studied areas of the Bay of Fundy and Gulf of St. Lawrence.

In offshore areas, sandy sea floor is relict, at least in the sense that contemporary processes only rework sediment and there is an insignificant supply of new sand. The dominant sediment features in water depths of less than 115 m were developed during Holocene transgression in the nearshore zone.

Mud accumulates in shelf basins and on the continental slope at rates of 0.2 to 1 m per 1,000 years. The source of this mud is unknown. Some is probably exported from the coastal zone: pollen samples indicate that significant amounts of suspended particulate matter are exported offshore from coastal bays and estuaries[25], and coastal zone budgets[12,23,26] also suggest that several tens of percent of the fine sediment eroded at the coastline is exported to the continental shelf. Other sediment in continental shelf basins is probably reworked from relict sediments on the shelf, but the relative importance of "new" and "reworked" sediment supply is unknown.

Depositional Processes

The style of depositional process may determine the rate at which a pollutant is buried beneath the sediment-water interface. Is sedimentation a slow continuous process, or is it highly episodic, with short periods of very rapid sedimentation interspersed with long periods when little sediment accumulates? In estuarine environments sedimentation may be quasi-continuous, but in many coastal environments (such as beaches) the bulk of the sedimentation is highly episodic and related to storms. On the continental shelf, sand motion also appears to be episodic, while the evidence in mud accumulating areas is contradictory. For example, graded silt to mud beds point to highly episodic deposition in Emerald Basin[11] while the microfossil record suggests continuous sedimentation.

Chemical Fluxes Through Bottom Sediment

Substantial changes in the chemical composition of marine sediments occur at or just below the seabed, particularly with the release or adsorption of labile trace metals. Many of the relevant reactions are highly redox dependant. In most areas off eastern Canada, the sediment surface is mildly oxidizing, becoming mildly to strongly reducing at a few centimeters depth. Physical disturbance of this redox gradient may greatly accelerate chemical reactions.

Detailed studies of geochemical fluxes through surficial sediments in eastern Canada are restricted to work in estuaries (summarized by Buckley and Winters[27] where trace metal concentrations are often high. Similar processes are inferred to occur in the deep shelf basins which also contain organic rich mud. Concentrations of labile trace metals are probably substantially lower in till, sands, and Emerald Silt equivalents.

CONCLUSIONS

This review demonstrates that we have a good reconnaissance level knowledge of the distribution of rock and sediment types offshore eastern Canada, that can serve as a geological basis for assessing the quality of the marine environment. However, we have a limited and generally qualitative understanding of the processes that modify, erode, and deposit sediment at the seabed. In consequence, we can make at best only qualitative statements about the fate of environmental contaminants that behave as part of the surficial sediment system.

REFERENCES

1. King, L.H. and B. MacLean. 1976. Geology of the Scotian Shelf. Geological Survey of Canada, Paper 74-31.

2. King, L.H. 1972. Relation of plate tectonics to the geomorphic evolution of the Canadian Atlantic Provinces. Geol. Soc. Am. Bull., 83: 3083-3090.

3. King, L.H. 1980. Aspects of regional surficial geology related to site investigation requirements—Eastern Canadian Shelf. *In:* D.A. Ardus (ed.), Offshore Site Investigation. Graham and Trotman, London, p. 37-60.

4. King, L.H. 1970. Surficial geology of the Halifax-Sable Island map area. EMR, Marine Sciences Branch, Paper 1, 16 p.

5. Quinlan, G. and C. Beaumont. 1981. A comparison of observed and theoretical postglacial relative sea level in Atlantic Canada. Can. J. Earth Sci., 18: 1146-1163.

6. Greenwood, B. and P.B. Hale. 1980. Depth of activity, sediment flux and morphological change in a barred nearshore environment. Geological Survey of Canada, Paper 80-10, 89-109.

7. Vandermeulen, J.H., D.E. Buckley, E.M. Levy, B. Long, P. McLaren and P.G. Wells. 1978. Immediate impact of AMOCO CADIZ environmental oiling: oil behaviour and burial and biological aspects. Centre National pour l'Exploitation des Oceans, Actes de Colloques, No. 6, p. 159-174.

8. Lewis, C.F.M. and J.V. Barrie. 1981. Geological evidence of iceberg groundings and related sea floor processes in the Hibernia discovery areas of the Grand Banks, Newfoundland. Proceedings of the symposium on production and transportation systems for the Hibernia discovery. St. John's, Nfld. pp. 146-147.

9. Hill, P.R. , D.J.W. Piper and W.R. Normark. 1983. Pisces submersible dives on the Scotian Slope at 63° W. Geological Survey of Canada, Paper, 83-1A, 65-69.

10. Petrie, B. and P. C. Smith. 1977. Low frequency motions on the Scotian Shelf and Slope. Atmosphere, Vol. 15, 117-126.

11. Kontopoulos, N. and D. J. W. Piper. 1983. Storm graded sand at 200 m water depth, Scotian Shelf, eastern Canada. Geomarine Letters, 2, 77-81.

12. Piper, D. J. W., P. J. Mudie, J. R. J. Letson, N. E. Barnes and R. J. Iuliucci. 1984. The marine geology of the South Shore, N.S. Geological Survey of Canada, Paper 84. (in press)

13. Swift, D. J. P. , B. R. Pelletier, A. K. Lyall and J. A. Miller, 1973. Quaternary sedimentation in the Bay of Fundy. Geological Survey of Canada, Paper 71-23, p. 113-151.

14. Greenberg, D. and C. L. Amos. 1982. Suspended sediment transport and deposition modelling in the Bay of Fundy, N.S. A region of potential tidal power development. Can. J. Fisheries and Aquatic Sciences, Vol. 40, Supplement 1.

15. Letson, J. R. J. 1981. Sedimentology of southwestern Mahone Bay, N.S. M.Sc. Thesis, Dalhousie University, Halifax, N.S., 199 p.

16. De Iure, A. 1983. The effects of storms on sediments in Halifax Inlet, Nova Scotia. M.Sc. Thesis, Dalhousie University, N.S., 237 p.

17. Risk, M. J. and R. K. Yeo. 1980. Animal-sediment relationships in the Minas Basin, Bay of Fundy. Geological Survey of Canada, Paper 80-10, p. 189-194.

18. Risk, M. J. , R. D. Venter, S. G. Pemberton and D. E. Buckley. 1978. Computer simulation and sedimentological implications of burrowing by *Axius serratus*. Canadian Journal of Earth Sciences, 15(8): 1370-1374.

19. Lewis, C. F. M., B. MacLean and R. K.. H. Falconer. 1980. Iceberg scour abundance in Labrador Sea and Baffin Bay: a reconnaissance of regional variability. Proc., 1st Can. Conf. Marine Geotech. Engineering, 79-94.

20. Jacques McClelland Geoscience Inc. 1982. Seabed stability on the continental shelf of eastern Canada. Report to D.S.S. for Atlantic Geoscience Centre. Released in part as Geological Survey of Canada Open File 878.

21. Doxsee, W. W. 1948: The Grand Banks Earthquake of November 19, 1929. Publications of Dominion Observatory, 7, 323-336.

22. Piper, D. J. W. and M. J. Keen. 1976. Geological studies in St. Margaret's Bay, N.S. Geological Survey of Canada, Paper 76-18, 18 p.

23. Amos, C.L. and G. H. E. Joice. 1977. The sediment budget of the Minas Basin, Bay of Fundy, N.S. Bedford Institute of Oceanography Data Series BI-D-77-3, 411 p.

24. Barrie, C. Q. and D. J. W. Piper. 1982. Marine geology of Makkovik Bay, Labrador. Geological Survey of Canada, Paper 82-17.

25. Mudie, P. J. 1980. Palynology of later Quaternary marine sediments, eastern Canada. Ph. D. Thesis, Dalhousie University, Halifax, N.S.

26. Amos, C. L. and B. F. N. Long. 1980. The sedimentary character of the Minas Basin, Bay of Fundy. Geological Survey of Canada, Paper 80-10, 153-180.

27. Buckley, D. E. and G. V. Winters. 1983. Geochemical transport through the Miramichi Estuary. Can. J. Fisheries and Aquatic Sciences. 40, Supp. 2, 162-182

RIVER INPUT OF POLLUTANTS TO THE WESTERN NORTH ATLANTIC

R. F. Addison*

Freshwater input to the Northwest Atlantic has recently been reviewed by Bugden *et al.*[1] and much of the following data has been taken from that document. Between 30° and 90° N total freshwater input to the North Atlantic is about 2347 km^3/yr, but over half of this lies between 45° and 55° N. The major input in this zone is the St. Lawrence River and its tributaries which together account for about 424 km^3/yr, or about 35% of the 45°-55° N sources. However, this 424 km^3/yr is equivalent to only about 3% of the total volume of water estimated to move outward through the Cabot Strait (13000 km^3/yr). In other words, the bulk of the water circulating around the Scotian Shelf and Gulf of St. Lawrence comes from "outside" and considering the western North Atlantic as a whole, probably less than about 5% of the water is identifiable as river input.

Pollutants have been detected and measured in the St. Lawrence and its tributaries. Metal concentrations in the system have recently been reviewed by Yeats and Bewers[2]. Very generally, metal concentrations (whether in dissolved or particulate form) are not dramatically higher than those in other rivers or in obviously uncontaminated marine samples. The exceptions to this general statement are Cd, Ni, Zn, Pb and Cr, all of which may be elevated. This elevation may represent global environmental contamination (especially in the case of Cd, Zn and Pb) as well as, or instead of, local contamination. There are other more dramatic examples of local contamination, the most notable being that of Hg pollution in the Saguenay system[3].

Relatively little information is available about the input of other pollutants via the St. Lawrence River. Various organochlorine compounds are reported in the water of the St. Lawrence or in its tributaries[4] but organochlorine levels in sediments of the river or in the Gulf are typical of background contamination levels[4,5], usually less than about 50 mg/kg. Biota in the Gulf also tend to have organochlorine levels which indicate some environmental contamination[6,7,8] but certainly not to the extent found in comparable biota from more obviously polluted areas such as the Baltic[9,10].

Taken together, these data lead to the conclusion that since freshwater runoff represents a relatively minor input to the western North Atlantic, and since pollutant concentrations in that runoff are generally low (with a couple of exceptions), river input does not represent the major source of pollutants to the area.

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REFERENCES

1. Bugden, G.A., B.T. Hargrave, M.M. Sinclair, C.L. Tang, J.C. Therriault and P.A. Yeats. 1982. Freshwater runoff effects in the marine environment: the Gulf of St. Lawrence example. Can. Tech. Rept. Fish. Aquat. Sci., No. 1078.

2. Yeats, P.A. and J.M. Bewers. 1982. Discharge of metals from the St. Lawrence River. Can. J. Earth Sci., 19: 982-992.

3. Loring. D.H. and J.M. Bewers. 1978. Geochemical mass balances for mercury in a Canadian fjord. Chem. Geol., 22: 309-330.

4. Couillard, D. 1982. Evaluation des teneurs en composés organochlorés dans le fleuve, l'estuaire et le golfe St. Laurent, Canada. Env. Poll. (B), 3: 239-270.

5. Cannone, P. and Mamarbachi. 1975. Residus des insecticides organochlorés dans les sediments du Haut Estuaire du Fleuve St. Laurent. Bull. Env. Contam. Toxicol., 14: 83-87.

6. Duffy, J.R. and D. O'Connell. 1968. DDT residues and metabolites in Canadian Atlantic coast fish. J. Fish. Res. Bd. Canada, 25: 189-195.

7. Addison, R.F., M.E. Zinck and R.G. Ackman. 1972. Residues of organochlorine pesticides and polychlorinated biphenyls in some commercially produced Canadian marine oils. J. Fish. Res. Bd. Can., 29: 349-355.

8. Addison, R.F., S.R. Kerr, J. Dale and D.E. Sergeant. 1973. Variation of organochlorine residue levels with age in Gulf of St. Lawrence harp seals *(Pagophilus groenlandicus)*. J. Fish. Res. Bd. Can., 30: 595-600.

9. Jensen, S., A.G. Johnels, M. Olsson and G. Otterlind. 1972. DDT and PCBs in herring and cod from the Baltic. Ambio Spec. Rept., 1: 71-85.

10. Helle, E., M. Olsson and S. Jensen. 1976. DDT and PCB levels and reproduction in ringed seals from the Bothnian Bay. Ambio, 5: 188-189.

CONTAMINANT TRANSPORT THROUGH THE MARINE ENVIRONMENT

E.W. Brandon* and P.A. Yeats**

The atmosphere is a carrier of anthropogenic and natural materials to the oceans. Trace metals, organic compounds, carbon dioxide, radioactive materials and micro-organisms all may enter the oceans via the atmosphere. Rivers also carry trace constituents to the ocean in both dissolved and particulate phases. These trace inputs are ultimately removed to the sediments by association with settling particles. In this chapter, we discuss the processes of atmospheric transport of pollutants to the Northwest Atlantic and the transport and removal processes that occur within the ocean.

ATMOSPHERIC DISPERSION

Turbulent dispersion in the lowest atmospheric layers depends on surface roughness and temperature and on vertical wind structure. Above the first few kilometres the influence of surface roughness becomes negligible. Figure 1(a) depicts a temperature inversion. Vertical dispersion is generally inhibited by such an inversion. The vertical wind structure may, on rare occasions, tend to offset the effect upon dispersion of the gradient of temperature in the vertical referred to in meteorological circles as the temperature lapse rate. However, detailed information about the vertical wind profile is usually missing.

Generally, mixing is enhanced in a layer in which the vertical temperature lapse rate is unstable; if there is no precipitation this usually means that there is a temperature decrease with height of more than 1°C/100 m. Such unstable profiles are usually capped by an inversion of some sort where further mixing in the vertical is inhibited, as shown in Figure 1(b). A temperature lapse rate of about 1°/100 m is termed "neutral" because vertical motions are neither enhanced nor damped, provided that the air is unsaturated.

The "boundary layer" is defined as that layer next to the surface of the earth where pollutants circulate in the vertical and horizontal as a result of small scale turbulence. Pollutants released into the boundary layer are confined to it, more or less uniformly mixed in the vertical dimension with it. This boundary layer is usually much thicker over land during the day than at night, and the diurnal change in boundary layer thickness is much greater over land than over water, where there is frequently no diurnal change at all. Boundary layer heights may range from almost zero to several kilometres. Over land, the turbulence responsible for creating a more or less

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Fig. 1a

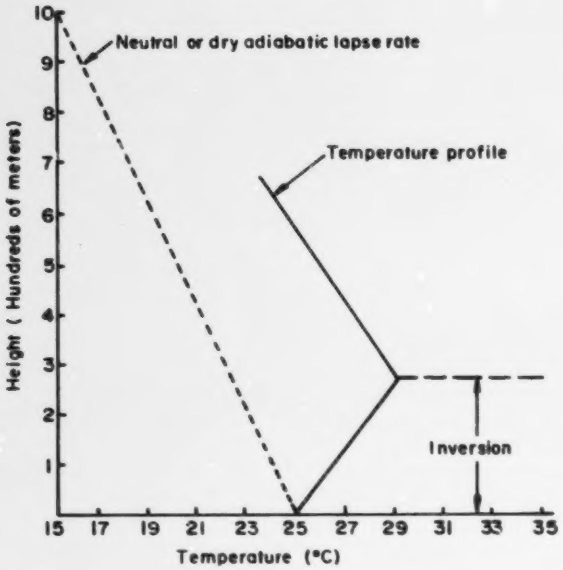

Fig. 1b

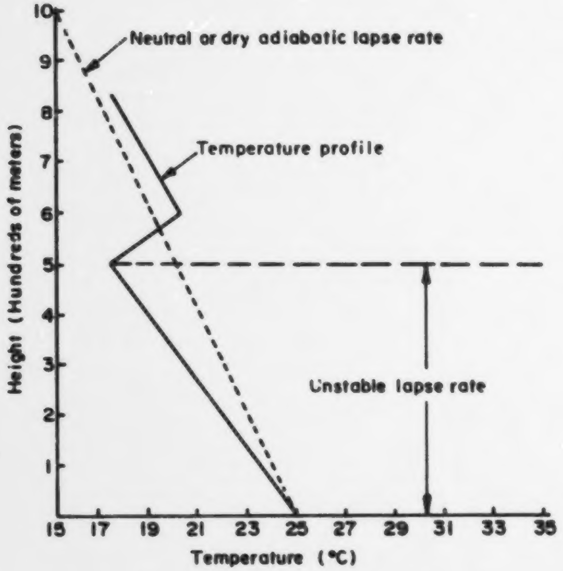

Figure 1: *(a) Surface temperature inversion below an unstable layer;*
(b) surface unstable layer capped by an inversion.

uniform dispersion of pollutants in the boundary layer dies out just after sunset. The mixed layer then becomes thinner and those pollutants formerly confined to it now become part of the "free" atmosphere. According to Deardorff[1] this mechanism is mainly responsible for the presence of pollutants above the boundary layer. Over the oceans, this mechanism is much less important for carrying pollutants aloft into contact with the free atmosphere.

The lift associated with the movement of colder air over a warmer water surface is probably a major mechanism for pollutant mixing over the open ocean. Convective clouds which occasionally reach the thunderstorm stage may form under these circumstances. Mixing throughout an extensive portion of the troposphere may take place along with such convective activity.

Synoptic scale features and high and low pressure areas are associated with organized patterns of vertical motion. Figure 2 shows schematically the vertical and horizontal movements of air parcels which are related to synoptic scale features. Air parcels bearing various pollutants may be lifted ahead of a travelling low pressure system. Since winds aloft are generally much stronger than close to the earth's surface, the lifted air parcels may be carried well ahead of the system finally descending to the west of the next downstream low pressure system. Thus, large scale pressure systems may bring about mixing to great heights in the troposphere.

ATMOSPHERIC TRANSPORT AND DEPOSITION

Materials may be transported long distances in the atmosphere. Figure 3 shows vector mean surface winds across the North Atlantic. The general westerly trend is clear. This trend is even more temporally homogeneous with height.

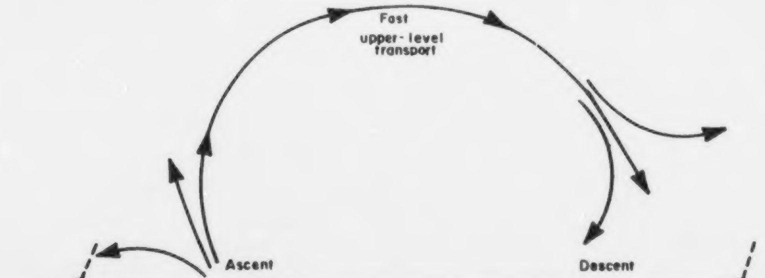

Figure 2: *Ascending and descending motions associated with synoptic scale low pressure areas.*

Recently disclosed environmental problems, such as that of acid rain, have stimulated research into atmospheric chemistry and trajectory modelling. Computer programs have been devised to take into account the three-dimensional structure of the atmosphere and the chemical changes which take place *en route*. These models are still fairly crude, and none are designed to reflect in detail the physics and chemistry of the real world. Most yield quite conservative results. However, such models may eventually permit accurate estimates of the atmospheric transport of pollutants from the land to the oceans. Machta[2] describes one such model, which although omitting many features of the real atmosphere, attempts to include atmospheric deposition.

Deposition at the ocean surface may be in the form of dry particles, through absorption of gaseous contaminants, or through "washout" (i.e., wet precipitation) of particulate material. An extension of Figure 4 (showing North Atlantic storm tracks) would show that most storms in the North Atlantic pass through the highly industrialized regions of North America. Subsequent precipitation from these storms therefore introduces pollutants, which may have originated in industrialized North America, into the North Atlantic.

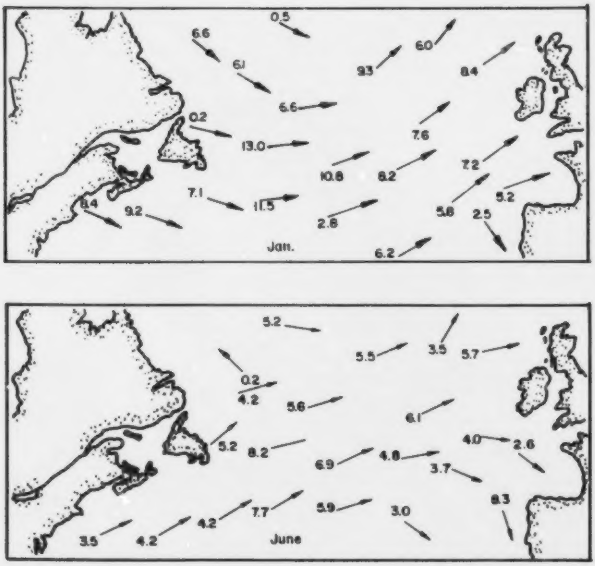

Figure 3: *Vector mean winds (knots) in the North Atlantic, in January and June*[3].

The relationships which govern the transfer of materials at the ocean surface are complex, and the unknowns are numerous. For example, sea spray may significantly affect the transfer of materials at the sea surface[4]. The rate at which material is deposited (deposition velocity) is defined as: amount of material deposited during unit time per unit area divided by the concentration of the material in question in the layer of air next to the surface.

Deposition velocities over the ocean are expected to be in the range 0.01-10 cm/sec for particles in the 0.1 µm range. These data apply to dry deposition. For wet deposition, few data relating to storm occurrences, duration or precipitation amounts over the open ocean are available.

MARINE TRANSPORT, DISPERSAL AND REMOVAL MECHANISMS

Transport and dispersal of chemical contaminants in the marine environment occur mainly through physical mixing and transport of the water. The action of wind and waves, tidal currents, coastal upwelling, estuarine mixing and entrainment processes, and any other mixing process will tend to disperse pollutants within the water column. On a larger scale, ocean currents will dominate the transport of chemicals introduced through

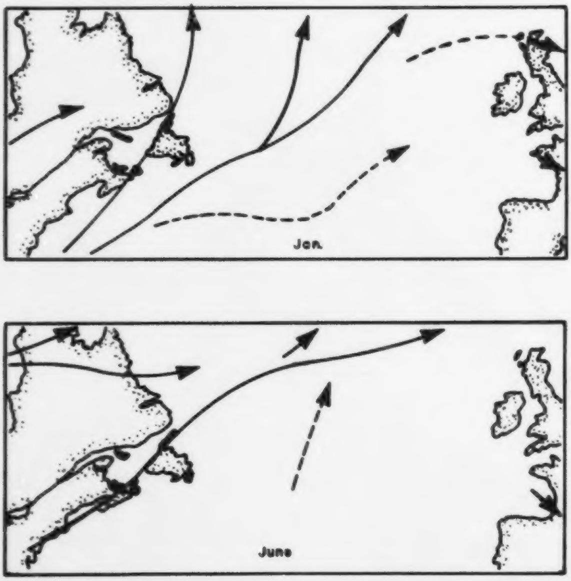

Figure 4: *North Atlantic storm tracks in January and June. Principal tracks, solid arrows; secondary tracks, dashed arrows*[3].

the atmosphere or by continental runoff. These processes have been discussed in previous chapters. Superimposed on these processes are the effect of geochemical transformation of chemicals within the marine environment. The first point to consider in discussing transport and dispersal of chemicals in the sea is the source of the materials. The two main sources of pollutants are atmospheric deposition (both wet and dry) and river discharge of both dissolved and particulate material. Other sources, such as discharge from ships or undersea volcanic activity, may occur but are of less general significance. The relative importance of these sources illustrated by calculated global fluxes of selected trace metals is shown in Table 1. The net particulate discharge from rivers is calculated assuming 5% of river-borne particulates escape the estuarine region. Dissolved material derived from river input may be mixed by several estuarine processes which result in dilution with sea water. The extent of mixing is most easily indicated by the salinity of brackish water. Simple dilution of river-borne chemical inputs will result in a linear relationship between the chemical of interest and salinity. Chemical reactions can either deplete or augment the fluxes of the dissolved species through the estuary, resulting in non-linear relationships with salinity.

The addition of dissolved components may arise from solution or desorption from particles, efflux from sediments, or dissolved phase reactions that produce the component. Reactions in the water column may occur as a result of changes in pH, ionic strength or oxygen concentration in estuarine waters. Efflux from the sediments will result from diagenetic changes in the sediment and will be accentuated by resuspension of the sediments. Reducing conditions in the sediments will often facilitate chemical remobilization. Taken together, these processes will result in non-linear pollutant-salinity relationships, often with maximum deviation from

TABLE 1 - GLOBAL METAL INPUT TO THE OCEAN
flux (x10^9g/yr)

Source of Input	Mn	Ni	Cu	Zn	Cd	Hg	Pb
Atmosphere	50	8	5	50	1	0.5	50
River (dissolved)	200	30	50	100	1	0.3	30
River (particulate)	21,000	1,200	1,400	2,300	6	3	600
Net particulate river flux*	1,000	60	70	110	0.3	0.1	30
Remobilization within coastal zone	5,200	200	400	300	0	0.4	200

* 5% of total particulate flux from rivers.

linearity at low salinity. This type of behaviour has been observed for several metals including Mn, Cu, Ni and Cd.

Estuarine removal may result from precipitation of dissolved components or flocculation of colloidal material. Adsorption or co-precipitation of trace components may often accompany these processes. Adsorption to ambient river-borne, marine or resuspended particulates can also occur. Removal reactions of this sort may be enhanced in that part of the estuary described as the "turbidity maximum" where suspended particulates may be elevated by an order of magnitude. This region is usually most developed near the head of an estuary where salinities may be in the 0-5°/oo range. Removal of dissolved components will be maximized in this region not only because of the high suspended particulate concentration, but also because organic flocculation will be maximized here and the most rapid changes in ionic strength will occur. These processes of estuarine removal are most evident in the case of iron where precipitation of the hydrated iron oxides or of organo-iron compounds are observed in almost all estuaries. Removal of metals such as Cr and Zn may accompany Fe precipitation.

As shown in Table 1, the dominant form of trace metal transport is generally in the particulate fraction. As many trace organic materials are hydrophobic, they will tend to be adsorbed to particulate surfaces and will be transported with particulate discharge. Most river-borne particulates are deposited in estuarine and nearshore sediments. However, several processes in these regions may reduce the depositional fluxes of trace particulate components. First, larger particles will be preferentially removed. Since metals and other contaminants are more concentrated in smaller than in larger particles (because of the larger relative surface area), this selectivity in deposition results in a disproportionately high metal discharge. Newly flocculated material, which can act as a sorbent for dissolved metals, may also be relatively slow to settle and may be selectively discharged.

As a result of the decomposition of organic matter and other diagenetic changes in estuarine sediments, a fraction of the particulate metals deposited in estuarine and nearshore sediments will be mobilized and returned to the water column. Decomposition of organic matter may also release trace organic components. The sediment may therefore act as a secondary source of material which was initially deposited as river-borne suspended matter. Once in the water column, these chemical traces may become associated with ambient fine grained particles, or they may remain in the dissolved phase, but in either case a large proportion will be transported offshore. Extensive remobilization of Mn by this process has been observed, and other metals such as Cu may be released in this way. Atmospheric inputs to the ocean are particularly significant in the cases of elements such as Fe and Al which are major components of terrestrial dust (and therefore not strictly pollutants) and for volatile trace elements such as Se, Hg, Pb and Cd, radioactive fallout products such as Cs-137 and Sr-90, and several transuranics. For Fe and Al, the atmospheric dust input is similar to the riverine dissolved input and in the case of Hg, Pb and Cd, atmospheric input may exceed riverine input. For trace organics such as DDT, atmospheric input may also exceed riverine input.

Chemical reactions involving species entering the ocean through atmospheric input are poorly understood. Some inputs, such as those of Cs-137 and Sr-90, will remain predominantly in the dissolved phase and can be transported by water movements. Others will be removed by adsorption to sinking inorganic or biogenic particles, or by active incorporation into marine biota. A third possibility, applying particularly to hydrophobic organic materials, is concentration in the surface micro-layer and a return to the atmosphere through evaporation or formation of aerosols.

Chemicals introduced to the ocean are transported horizontally by ocean currents and ocean mixing and vertically by advection (upwelling and deep water formation), turbulent mixing and settling of particulate matter. Since physical circulation and mixing processes have been discussed elsewhere, we will concentrate on the effect of particulate transport on the removal of trace chemical constituents in the oceans. However, it should be emphasized that for the removal of most trace constitutents from surface water, vertical advective transport and mixing are of similar magnitude to the particulate fluxes, and horizontal transport is almost totally controlled by physical processes.

The most important ultimate removal process is incorporation of contaminants to settling particles. Other mechanisms, such as direct adsorption to sedimentary surface, may play a role, but a less important one. Two distinct types of particulate matter are significant: terrestrial organic and inorganic particulates transported to the ocean by rivers or directly by atmospheric fallout, and biological material produced *in situ*. Mineral formation within the ocean is likely to be important only in the case of MnO_2 precipitation.

The settling of river-borne particulates in estuaries is the obvious removal mechanism for most particulate chemical input from rivers. Precipitation, flocculation and adsorption may contribute to this removal. Further offshore, river-borne particulate settling becomes less important and settling of atmospheric particles more important. As particles settle through the water column, dissolved constituents will also be adsorbed or otherwise scavenged by these particles. In some cases, such as MnO_2 precipitation, the presence of particulate surfaces can catalyse inorganic precipitation reactions. These inorganic processes are mainly responsible for the vertical distributions and removal of elements such as Al, Cu and Pb.

The uptake of metals by biota, and their subsequent removal from the water column by planktonic debris or fecal pellets, controls the distribution and transport of several trace metals. This process may also be significant for many organic contaminants. Biological removal controls the vertical distribution of Cd, Zn, Ni, Cr and Se which have classical nutrient-like profiles with low levels in surface waters and maxima coinciding with the phosphate or silicate maxima. Some other metals such as Fe, Al and Cu may also be involved in these biogeochemical processes but to a lesser extent. Biota may either take up these metals actively and incorporate them into skeletal or other tissue, or passively by adsorbing them to external surfaces. In either case, biological activity has the net effect of removing metals from surface waters to intermediate depths where maximum net biological regeneration may occur. A fraction of this biological debris, with its associated trace metals, will be transported through the water column to sediments. As shown in Table 2, such transport may be quite large. However, most of the biologically removed material is regenerated in subsurface and intermediate waters and returned to the surface by physical processes. A small fraction of the biogenic debris escapes to the sediments and represents the major mechanism of metal removal from the surface to the ocean floor.

Large amounts of inorganic and organic matter are processed by zooplankton during feeding and are excreted as fecal pellets. This material is quite dense and settles rather quickly, taking with it appreciable amounts

TABLE 2 - REMOVAL OF METALS FROM OCEANIC SURFACE WATER, flux (x10^{9}g/yr)

Form of Flux	Mn	Ni	Cu	Zn	Cd	Hg	Pb
Water mixing and advection	500	600	200	200	40	9	70
Settling of terrigenous particles	800	50	90	100	0.3	0.1	20
Settling of biogenic soft parts	0	700	200	0	500	8	30
Settling of biogenic skeletal fragments	100	700	300	2,000	4	3	80
Settling of fecal pellets*	200	200	200	800	8	0.3	30
Adsorption and authigenic mineral formation	5,000	80	0	0	0	0.8	50

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* Fecal pellet flux will include skeletal fragments and terrigenous particles incorporated into the fecal pellets.

of trace metals. Since zooplankton ingest particulate matter rather indiscriminately and discard unwanted components in their feces, inorganic and organic materials not essential to biological processes can be transported to the sediments in this way.

Organic contaminants and radionuclides may also be removed to the sediments by reaction with particles. In the case of organic contaminants, interaction with biological material seems likely to be a major factor in controlling their distributions. Fallout radionuclides such as Cs-137 and Sr-90 are fairly soluble in sea water and are transported conservatively, but their ultimate removal is probably by association with particulate material. Less soluble radionuclides such as those of thorium and plutonium should be more rapidly associated with particles and may be expected to behave in sea water like the trace metals such as Pb and Cu (scavenged by inorganic particles) or Cd and Zn (removed by biogenic particles).

A final method of removal of contaminants from sea water involves transfer from the surface to the atmosphere by evaporation or by aerosol formation. This is the major route of removal of Cl^- from sea water, and may apply to some trace components. Volatile organics and possibly some volatile inorganics such as I and Hg may be significantly evaporated from the surface. Hydrophobic organics which are concentrated in the microlayer may be incorporated into aerosols. However, good quantitative data to support these suggestions are lacking.

Some Examples

Carbon Dioxide

There is about sixty times more CO_2 in the sea (mostly as carbonate and bicarbonate) than there is in the atmosphere. Its solubility in sea water decreases rapidly with temperature. Long term climatic variations may be associated with unstable interactions between biological evolution and the cycling of CO_2 and O_2 through the global system[5]; cyclical changes in atmosphere CO_2 associated with the overturning of the deep oceans on a time scale of a few millenia may contribute to the glacial-interglacial cycle. On shorter time scales, CO_2 produced anthropogenically may have a dramatic influence on the environment. Although CO_2 was not in the past considered a harmful pollutant, the present rate of production of CO_2 through burning could bring about some climatic changes in as little as fifty years (this is approximately the time frame in which sulphur oxides and other pollutants have produced changes to our environment). The ocean is an important potential sink for excess CO_2, but as the ocean takes several hundred years to mix, it cannot absorb all the excess CO_2 produced in the next few decades. From the point of view of ocean contamination, some authors suggest that the excess CO_2 absorbed in the surface layers might produce a sufficiently large change in pH, and hence $CaCO_3$ solubility, that marine animals may not be able to form shells. However, this appears not to be the majority view of scientists working in the field[6].

Trace Metals

Although most metals enter the ocean largely through rivers (see Table 1 for flux estimates), several metals have significant atmospheric sources. Of particular significance are those metals released during the burning of fossil fuels, including Se, whose total release is estimated[7] to be about 7×10^9 g/yr and which may attain air concentrations over the North Atlantic in the range of 0.09-0.4 ng/m^3, and V (derived from petroleum) which may reach atmospheric concentrations in northern hemisphere westerlies[8] of around 0.2 ng/m^3. Metals entering the atmosphere from industrial sources include Pb, which may enter the ocean from the atmosphere at about forty times the "natural" rate of input from other sources[9]; Hg, whose atmospheric concentration is around 1-2 ng/m^3, and which enters the ocean mainly from atmospheric sources[10]; and Cd, whose atmospheric concentrations range from 0.9 ng/m^3 to the µg/m^3 range in the vicinity of Zn smelters[11]. All these metals have atmospheric fluxes that are comparable to their river inputs in the dissolved phases.

Organochlorine Compounds

Both the DDT group of insecticides and the PCBs are appreciably volatile at normal environmental temperatures, and are found in the North Atlantic ocean atmosphere in both gaseous and adsorbed forms[12]. That they can be transported over long distances is shown by the data of Seba and Prospero[13], who found DDT group residues in the Caribbean atmosphere which was derived from insecticide spraying activities in West Africa, 4,500 km to the east. The significance of atmospheric input of these materials to the ocean is not known quantitatively, except that it is large. In the western North Atlantic, there is indirect evidence that atmospheric input may be the major source of PCBs[14]. From studies of DDT group residues in rainwater and in other environmental samples from New Brunswick, Pearce *et al.*[15] concluded that about one-quarter of the DDT sprayed over New Brunswick in 1968 was transported to the Gulf of St. Lawrence by atmospheric transport.

More discussion of the sources and transport of organochlorines to the sea will be found elsewhere in this document.

REFERENCES

1. Deardorff, J.W. 1975. Mixing through the boundary layer. A Workshop on the Tropospheric Transport of Pollutants to the Ocean. Miami, Fla., U.S.A.

2. Machta, L. 1975. A computer program for estimating pollutant transfer from air to sea. A Workshop on the Tropospheric Transport of Pollutants from the Continents to the Oceans. Miami, Fla., U.S.A.

3. Meserve, J.M. 1974. U.S. Navy Marine Climactic Atlas of The World. Vol. I. North Atlantic Ocean (revised ed.). Naval Suc. detachment, Ashville, N. Carolina.

4. Hicks, B.B. 1975. Some notes regarding the air/sea exchange of pollutants. A Workshop on the Tropospheric Transport of Pollutants from the Continents to the Oceans. Miami, Fla., U.S.A.

5. Kraus, E.B. 1972. Atmosphere - ocean interaction. Clarendon Press, Oxford.

6. Jones, E.P. 1979. An overview of long range transport of air pollutants and the oceans. Proceedings of a Workshop on Long Range Transport of Air Pollution and Its Impacts on the Atlantic Region. Dartmouth, Nova Scotia, October 17-18, 1979.

7. Chester, R. 1975. Arsenic and Selenium. A Workshop on the Tropospheric Transport of Pollutants to the Ocean. Miami, Fla., U.S.A.

8. Duce, R.A. 1975. Atmospheric vanadium transport to the ocean. A Workshop on the Tropospheric Transport of Pollutants to the Ocean. Miami, Fla., U.S.A.

9. Patterson, C.C. 1975. Transport of industrial lead to the seas. A Workshop on the Tropospheric Transport of Pollutants to the Ocean. Miami, Fla., U.S.A.

10. Goldberg, E. 1975. Atmospheric transport of mercury from the continents to the oceans. A Workshop on the Tropospheric Transport of Pollutants from the Continents to the Oceans. Miami, Fla., U.S.A.

11. Peirson, P.H. 1975. Cadmium. A Workshop on the Tropospheric Transport of Pollutants from the Continents to the Oceans. Miami, Fla., U.S.A.

12. Eidleman, T.F. and C.E. Olney. 1974. Chlorinated hydrocarbons in the Sargasso Sea atmosphere and surface water. Science 183: 526-528.

13. Seba, D.B. and J.M. Prospero. 1971. Pesticides in the lower atmosphere of the northern equatorial Altantic Ocean. Atmospheric Environment, Vol. 5: 1043-1050

14. Ware, D.M. and R.F. Addison. 1973. PCB residues in plankton from the Gulf of St. Lawrence. Nature, 246: 219-251.

15. Pearce, P.A., L.M. Reynolds and D.B. Peakall. 1978. DDT residues in rainwater in New Brunswick and estimate of aerial transport of DDT into the Gulf of St. Lawrence, 1967-68. Pest. Monit. J. 11(4): 199-204

RADIOACTIVITY IN THE MARINE ENVIRONMENT

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INTRODUCTION

The first significant releases of radioactivity into the marine environment were from the Hanford atomic plant into the Pacific Ocean via the Columbia River in 1944. Releases of radioactivity on a global scale began with the detonation of two nuclear devices at Bikini Atoll in the Pacific Ocean in 1946. During the subsequent 35 years man has produced a quantity of a radioactivity which exceeds that which existed naturally in the lithosphere before 1942. Artificial radionuclides were the first group of chemical substances widely recognized to be a significant man-made perturbation to the chemistry of the ocean and subsequent studies of marine radioactivity have sensitized marine chemists to the threat to the ocean's resources posed by potential widespread contamination of the sea. In this paper the sources and nature of radioactive contamination of the western North Atlantic Ocean are outlined and the implications of future radioactive inputs to the environment are considered.

SOURCE OF RADIOACTIVITY

Three types of artifically produced radionuclides have been introduced into the marine environment: (1) nuclear fuels such as U-235 and Pu-238; (2) fission products, such as Cs-137 and Sr-90, arising from the use of nuclear fuels or nuclear weapons tests; and (3) activation products, such as Fe-55 and Zn-65, produced from the interaction of nuclear particles with the components of nuclear fuel and weapons. Artifically produced radionuclides which have been detected in the marine environment are listed in Table 1. These include elements which emit alpha and beta particles and gamma rays and include radionuclides with a wide spectrum of half lives ranging from approximately 1 day for W-187 to 210,000 years for Tc-99.

By far the greatest contribution to the global inventory of radioactivity has resulted from nuclear weapons tests conducted in the atmosphere by the U.S.A. and the U.S.S.R. and to a lesser extent by the U.K., France, India, and the People's Republic of China. Since 1945 there have been about 800 nuclear detonations having a total yield of about 325 megatons[2]. Much of the nuclear debris from these explosions was initially distributed by meteorological processes throughout the stratosphere and was later deposited on the earth's surface. Since almost all nuclear weapons testing was conducted north of the equator, much of it by the U.S.S.R. at 75° N in the Arctic, most (approximately 80%) of the fallout occurred in the northern hemisphere. The bulk of this radioactivity input occurred during the 1950's and 1960's, and the global inventory of fallout radioactivity has declined significantly since the cessation of widespread nuclear weapons tests in 1963.

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Nuclear reactors also generate large quantities of radioactivity, but, to date, there have been few instances of significant releases of radioactivity from this source. The sole Canadian coastal reactor is a 660 Mw CANDU reactor situated at Point Lepreau, New Brunswick, on the Bay of Fundy. Environmental effects associated with the operation of this reactor which began operations in 1982 are documented in environmental surveillance reports produced by the Atlantic Environmental Radioactivity Unit

TABLE 1 - ARTIFICIALLY PRODUCED RADIONUCLIDES WHICH HAVE BEEN DETECTED IN THE MARINE ENVIRONMENT[1]

Radionuclide	Half-Life*		Type of Decay**	Radionuclide	Half-Life		Type of Decay
H-3	12.26	y	β-	C-14	5.76×10^4	y	β-
Sr-89	51	d	β-	P-32	14.3	d	β-
Sr-90	28	y	β-	S-35	87.2	d	β-
Y-90	64.2	h	β-	Ca-45	165	d	β-
Y-91	59	d	β-	Sc-46	84	d	β-,γ
Nb-95	35	d	β-,γ	Cr-51	27.8	d	K,γ
Zr-95	65	d	β-,γ	Mn-54	314	d	K,γ
Mo-99	67	h	β-,γ	Fe-55	2.7	y	K
Tc-99	2.1×10^5	y	β-	Fe-59	45	d	β-,γ
Ru-103	40	d	β-,γ	Co-57	270	d	K,γ
Ru-106	1	y	β-,γ	Co-58	71	d	K,β+,γ
Sb-125	2.7	y	β-,γ	Co-60	5.26	y	β-,γ
I-131	8.04	d	β-,γ	Zn-65	245	d	K,β+,γ
Te-132	78	h	β-,γ	As-76	26.5	h	β-,γ
Cs-137	30.2	y	β-,γ	Ag-108m	127	y	K,γ
Ba-140	12.8	d	β-,γ	Ag-110m	253	d	β-,γ
La-140	40.2	h	β-,γ	Cd-113m	43	d	β-
Ce-141	32.5	d	β-,γ	Cd-115m	43	d	β-,γ
Ce-144	285	d	β-,γ	Sb-124	60	d	β-,γ
Pm-147	2.6	y	β-	Cs-134	2.1	y	β-,γ
Eu-155	1.81	d	β-,γ	W-181	30	d	K,γ
				W-185	73	d	β-
				W-187	24	h	β-
				Bi-207	28	y	K,γ
				Np-239	2.35	d	β-,γ
				Pu-238	86	y	α
				Pu-239	2.44×10^4	y	α
				Pu-240	6.6×10^3	y	α
				Pu-241	13.2	y	β-
				Am-241	458	y	α,γ
				Cm-242	163	d	α

* y, years; d, days; h, hours.
** β-,,beta particle; β+, positron;
γ, gamma ray; K, electron capture;
α, alpha particle.

(AERU) at the Bedford Institute of Oceanography. Although few in number, plutonium reprocessing plants, which extract plutonium from spent nuclear fuel, have released significant quantities of radioactivity to the oceans. The Sellafield (formerly, Windscale) reprocessing plant, located on the west coast of England, has released large quantities of Cs-137 into the Irish Sea since the early 1970's and a French reprocessing plant at Cap de La Hague has also discharged significant quantities of fission products and transuranic radionuclides into the North Sea. However, the greatest potential for future radioactive contamination of the oceans from the nuclear fuel cycle is the disposal of radioactive wastes in the sea. Low level dumping of nuclear waste has been under way at sites in the eastern Atlantic Ocean for the past 35 years. Several nations are seriously considering the feasibility of disposing of high level wastes by incorporating them into corrosion resistant, solid matrices and burying these in canisters below the sediment surface in geologically inactive regions of the ocean floor. Releases of radioactivity from disposal sites followed by biological uptake in coastal regions could lead to human exposure to radioactivity in maritime nations.

The only substantial global dissemination of radioactivity from sources other than nuclear weapons tests resulted from the accidental combustion of a SNAP-9A (Systems for Nuclear Auxiliary Power) nuclear powered navigational satellite. This occurred in 1964 when the satellite failed to reach orbital velocity because of a rocket malfunction shortly after launch, and burned up 46 kilometres above the Indian Ocean. Seventeen kCi of Pu-238 were released into the stratosphere, 95% of which had been deposited on the earth's surface by the mid-1970's, where it is easily measurable in sediments and biological phases. Radioisotopes are also encapsulated for uses in energy production, in food preservation, in the sterilization of medical equipment and in industrial thickness gauges. Only in the case of rupture of containers or careless handling and disposal of these materials is there a possibility of oceanic entry.

Cs-137 AND Sr-90 DISTRIBUTIONS

The greatest effort devoted to the study of oceanic distributions of artificially produced radionuclides has focused on measurements of Cs-137 ($t_{1/2} = 30$ yr) and Sr-90 ($t_{1/2} = 29$ yr), both of which were produced during nuclear weapons tests. Both radionuclides become isotopically diluted with their stable forms which exhibit conservative behaviour in seawater, so there has been great interest in them as tracers for the mixing of surface waters, particularly with deeper oceanic water masses. The pattern and time scale of Sr-90 deposition has been followed particularly closely because of its potentially deleterious biological impact (it follows the same metabolic pathways as calcium) and the existence of relatively direct pathways by which it can reach man. Cs-137 has also received increased attention in recent years because of its high radiotoxicity and its use as a particle tracer.

Sr-90 tends to follow the hydrological cycle in freshwater environments, while, in contrast, Cs-137 is strongly bound to fine grained clays and organic matter and tends to follow particle transport pathways through freshwater and estuarine systems. In seawater, both radionuclides exist mainly in their ionic states and remain predominantly in solution, although there is some evidence for the preferential removal of Cs-137 onto suspended particulate matter and sediments. The ratio of Cs-137/Sr-90 in fallout debris has remained relatively constant[3] at a value of about .15 and measurements of this ratio in large numbers of samples of open ocean water collected throughout the northern hemisphere show no significant deviation from this value[4,5]. Cs-137 and Sr-90 depth profiles measured off Newfoundland in 1972 are typical of those measured in North Atlantic waters; elevated activities[6] of the order of 7.4 mBq/L are found in the surface mixed layer, decreasing rapidly to levels of the order of 0.17 mBq/L below 1000 m. Measurements of Cs-137 and Sr-90 performed in the author's laboratory on surface water samples collected in Baffin Bay in 1981 have a Cs-137/Sr-90 ratio of 1.5. In nearshore waters this ratio is reduced to a value of about 1 due to preferential sorption of Cs-137 onto particulate material and consequent removal from solution in freshwater regimes. Many of the seawater samples collected in the Bay of Fundy between 1979 and 1981[7] had Cs-137/Sr-90 ratios in the range of 1.0-1.5. Freshwater fractionation between Cs-137 and Sr-90 is most evident in their extremely low ratios (<0.1) in the Great Lakes[8] and in the Winnipeg River (<0.5) during the 1970's[9].

The total inventories of Cs-137 and Sr-90 in the North Atlantic Ocean are 1.2×10^{17} and 8.4×10^{16} Bq, respectively[6]. Direct fallout is the dominant input mode for these radionuclides but ocean current transport and river continental runoff also contribute. The greatest advective inputs to the North Atlantic occur via the East Greenland Current and via the current though the Canadian Archipelago, with additional inputs due to overflow at the Denmark Strait and at the Iceland Scotland ridge. The greatest outflow occurs into the Norwegian Sea. These data have been used to estimate a mean sinking rate of near surface water to depths below 1000 m of close to 14×10^6 m^3/s during the 20 years prior to 1972[6].

Until recently, then, the major portion of radionuclide inputs to the Atlantic Ocean has probably been from stratospheric fallout. However, during the past decade, releases of Cs-137 and, to a lesser extent, Sr-90 in the low level liquid waste stream from the British Nuclear Fuels Limited, Sellafield (Windscale) reprocessing plant on the Irish Sea have significantly changed the fallout radionuclide inventory in the North Atlantic Ocean. The quantity of Sr-90 released from this source through 1979 exceeds the inventory measured in the North Atlantic north of 60° N in 1972, and the amount of Cs-137 released is approximately six times greater than the Cs-137 fallout inventory measured in the same region[10,11]. Clearly, the diminishing inputs of fallout Cs-137 to the Atlantic Ocean are being rapidly exceeded by the increasing inputs of Sellafield Cs-137. If this trend continues, the Cs-137 distribution in the North Atlantic Ocean through the latter part of the 20th century may be governed by releases from Sellafield.

The Sellafield effluent stream also carries "elapsed time indicators" which can be used to determine water mass transport times; these are isotope pairs with different half-lifes such as Cs-134 and Cs-137. Abrupt changes in the sources or ratios of specific nuclides such as Cs-137 and Sr-90 also yield time dependent information on water circulation patterns. The bulk of this effluent stream moves northward from its release point around the Scottish coast and then moves rapidly across the North Sea and into the northward-flowing Norwegian coastal current. Windscale effluent may therefore be viewed as a continuous jet of tracer radionuclides extending into the Norwegian, Greenland and Barents Seas from which point further injection into the Arctic and high latitude North Atlantic Oceans may be expected. Recent measurements by Livingston *et al*[12] at the LOREX ice station located close to the North Pole indicate that seawater labeled with Sellafield Cs-137 is found at a water depth of 1500 m and has apparently reached this location in less than 8 to 10 years. Measurements conducted in the author's laboratory on water samples collected in 1981 at the FRAM III ice station, northeast of Greenland in the Arctic Ocean, appear to confirm the appearance of Sellafield Cs-137 at a water depth of 1500 m. Clearly, the properties and magnitude of the Sellafield radionuclide releases make them promising as oceanographic and geochemical tracers for processes in the Arctic and North Atlantic Oceans during the next decade.

Cs-137 is also a valuable tracer for particle transport processes in several different Canadian coastal and estuarine environments. Measurements of the Cs-137 distribution, with other particle tracers such as Pb-210, have been used to estimate sedimentation rates and sediment geochronologies in the Bay of Fundy[7,13] and in the Saguenay Fjord, Quebec[14]. The unusually well resolved, time dependent flux of Cs-137 to the sediments of the Saguenay Fjord has provided the basis for a drainage basin model for this sytem which can be used to estimate soil residence times for chemical substances introduced from the atmosphere[15]. The distribution of Cs-137 has also been measured in continental slope sediments off Newfoundland, where penetration of this radionuclide occurs to depths of 6-10 cm in the sediment as a result of bioturbation[16].

FISSION AND ACTIVATION PRODUCTS

A number of fission products other than Sr-90 and Cs-137 have been detected in the marine environment. The distributions of the lanthanide nuclides of moderate half-life, Ce-144 and Pm-147 have been reported by Sugihara and Bowen[17] in various parts of the Atlantic Ocean. The depletion of these radionuclides in surface waters is probably due to sinking of the particles with which they are associated. Following weapons explosions, shorter lived nuclides such as Ru-103, Ce-144, Ce-141, Zr-95 and Nb-95 have been measured in ocean waters receiving long range fallout[18]. The rapid uptake of many of these radionuclides in coastal sediments is due to active scavenging by particulate matter which leads to fractionation of these elements both from one another and with respect to conservative tracers such as Sr-90.

Concentrations of Fe-55, a neutron activation product, in fish tissues have been measured quite extensively. Relatively high specific activities (ratio of the radioactive nuclide to the stable nuclide concentration) can occur in such tissues, probably reflecting the low concentrations of biologically available stable iron in surface waters and the comparatively ready uptake of the freshly introduced radioactive form. Much of the total fallout of Fe-55 was produced by explosions during 1961/62. Since then, there have been marked variations in Fe-55 concentrations in fish with time and latitude, as well as with feeding habits. A pronounced latitudinal dependence of levels in North Atlantic cod was found in 1967/69; the maximum specific activity of 8.2 Bq/mg Fe compared with a peak value of 1.1×10^3 Bq/mg Fe for Pacific salmon in 1964. Concentrations in tuna and salmon in late 1965 were about two orders of magnitude higher than those measured after earlier series of weapons tests[19,20]. Fe-55 measurements in seawater and sediment samples from the Atlantic Ocean led to an estimate of 10 years for the residence time for iron in open ocean water[21]. However, the usefulness of this tracer during the 1980's will be considerably reduced owing to its short half-life and the absence of recent inputs to the oceans.

CARBON-14 AND TRITIUM

Carbon-14 (C-14) and tritium (H-3) are two artifically produced radionuclides which have naturally occurring counterparts produced by the cosmic ray fragmentation of atmospheric gases. The distribution of these two radioactive tracers in the world's oceans can be used to resolve the time scales for water mass mixing and transport phenomena. Unlike most other fallout radionuclides, which are at least partly associated with particulates, C-14 and tritium follow the global pathways for carbon and water, respectively. The global input of C-14 has been constant for the past several thousand years, resulting in significant radioactive decay in C-14 ($t_{1/2} = 5280$ yr) in the deep water components of the ocean due to the long time scale for abyssal circulation. C-14 levels in the surface waters of the ocean have been slightly diluted by the addition of CO_2 from fossil fuel inputs during the 20th century, but this effect has been recently reversed by the addition of radiocarbon produced by nuclear weapons tests during the 1950's and 1960's. Thus, a sharp transient of anthropogenic C-14 has appeared in oceanic surface waters which has been used to establish the rate of exchange between surface water and the deep oceanic reservoirs[22,23].

In contrast to C-14, pre-nuclear age levels of tritium in the ocean were comparatively low. Tritium levels increased 100-fold as a result of nuclear weapons tests conducted in 1962 and 1963, nearly all as tritiated water. With a half-life of 12.26 years, the tritium transient in the surface and "younger" waters of the oceans provides a powerful probe for dynamic studies of circulation processes over time periods of 50 years.

Some profiles of H-3 have been measured in the Atlantic Ocean off Canada[24]. These data clearly show a plume of tritiated water, which marks the course of newly formed deep water from its origin in the Norwegian and Greenland Seas and its subsequent transport over the sill of the Denmark Strait towards the deep Atlantic Ocean[25].

Surface seawater samples from the open Atlantic Ocean collected in 1954 showed[26] tritium levels as high as 0.43 Bq/L. Closer to the North America coastline, tritium activities from 0.55 to 3.88 Bq/L were found in samples collected just 80 km southeast of Halifax, Nova Scotia. Tritium concentrations in North Atlantic surface water peaked in 1964/65 and have since decreased by a factor of more than five[27]. Between 1963 and 1967 the tritium activity range in Atlantic coastal water[28] was between 3.4 and 47.8 Bq/L. Increased tritium activities at coastal stations are due both to lower dilution factors by oceanic water vapour and to a higher activity of re-evaporated continental moisture. Tritium levels in the early 1970's were around 1 Bq/L in Atlantic Ocean surface [29] water off Newfoundland. Tritium levels in the Bay of Fundy in 1979/1980[7] were of the order of 1-2 Bq/L.

Inputs of tritium to the North Atlantic Ocean by air/sea vapour exchange are four times that by direct precipitation. River runoff and net tropospheric flow from the continents may contribute 10 to 15% respectively to the oceanic tritium inventory[30]. Of the approximately 400 kg of tritium that were injected into the earth's atmosphere from the testing of nuclear weapons in the 1960's, approximately one-half had decayed by 1972, and less than 5 kg still remained in the atmosphere. The rest resides in groundwater aquifers and in the oceans, with approximately 59 kg residing in the North Pacific and 66 kg in the North Atlantic Oceans. Since the surface area of the North Pacific exceeds that of the North Atlantic by a factor of 2, the similar inventories of tritium seems surprising. The discrepancy arises mainly from differences in evaporation and precipitation in both oceans, from the limited northward extension of the North Pacific, and from differences in continental runoff contributions which are about three times greater in the North Atlantic[29,31,32].

PLUTONIUM

Approximately 15,000 GBq of Pu-239, 240 have been released to the environment as the result of the large thermonuclear weapons tests conducted in the 1950s and 1960s[33]. The major portion of the plutonium in seawater becomes rapidly associated with particulate material, and in shallow water regions greater than 95% of the plutonium inventory is found in the sediments. Distribution coefficients, K_d (activity per gram of sediment activity per ml of water)[34] may range from 10^4 to 5×10^5. Even higher values of K_d (approximately 10^6 in the Bay of Fundy in 1980)[35], are generally measured for the distribution of plutonium between suspended particulate matter and seawater; these are typical of coastal environments. The behaviour of plutonium and other transuranic elements is complicated as they exist in several different oxidation states. Plutonium is predominantly in the Pu (V and VI) oxidation state in the waters of the Irish Sea, while in the interstitial waters of the underlying sediments it is in the lower Pu (III and IV) oxidation states[36]. The uptake of plutonium by sediments is a redox as well as an adsorption process. The adsorption of plutonium onto the surfaces of particles is reversible[34], consistent with Goldberg's hypothesis that the partitioning in the ocean of trace elements such as plutonium is controlled by an equilibrium distribution process[37].

Plutonium is depleted with respect to the conservative tracers, Cs-137 and Sr-90, in the surface waters of the Atlantic[38] and Pacific[39]. Thus results indicate that the residence time for plutonium in the water column is significantly shorter than that of Cs-137 and Sr-90 and that the depletion of plutonium from surface waters is due to the settling of particles through the water column. However, an unusual and stable maximum in the ratio of Pu-239, 240/Sr-90 has been observed between 400 m and 700 m below the surfaces of both the Atlantic and Pacific Oceans[5]. This suggests that plutonium is taken up by the surfaces of autochthonous particles which are incorporated into large faecal pellets; these particles gradually dissolve as they fall through the water column. The maximum in the plutonium concentration could result from reoxidation of Pu(IV) to Pu(V) as the settling particles are remineralized. The net result is that as plutonium is released back into the water column at depth, there is an inadequate supply of particles to permit significant reattachment onto surfaces, and, thereafter the redistribution of plutonium is governed by the same advection and diffusion processes that control the distribution of Cs-137 and Sr-90.

Plutonium is therefore a useful tracer for studying sedimentation and sediment redistribution by bioturbation. The detailed time dependent distribution of Pu-239, 240 in the sediments of the Saguenay Fjord, Quebec, has led to estimates of the residence times for fallout plutonium in the water column of the fjord, and in the soils of the drainage basin[15]. Pu-239, 240 has also been measured in the sediments of the Bay of Fundy where the ratios, Pu-239, 240/Cs-137 are consistent with those predicted for a bioturbated, marine environment[13]. From their measurements of the ratio, Pu-239, 240/Cs-137 in continental slope sediments in the western North Atlantic Ocean, Livingston and Bowen[40] argued that plutonium was being remobilized in the sediments and escaping back into the water column. If this is true, it has important environmental implications for the sub-seabed disposal of radioactive wastes containing the long-lived transuranic radionuclides. Edgington[34] has argued that Livingston and Bowen's results can be more simply explained by a combination of bioturbation and preferential diffusion of Cs-137 with respect to plutonium and that the extent of plutonium remobilization from these sediments may be small. However, it is clear that some remobilization of plutonium occurs, possibly promoted by oxidation of Pu(IV) to the Pu(V) oxidation state, and that significant return of plutonium from the sediments to the water column can occur in certain environments, as has been demonstrated by Noshkin and

Wong[41] at Enewetak in the Pacific Ocean. How general this process is remains to be established.

The uptake of plutonium by biological materials is an important factor in assessing the potential for this highly toxic and long-lived element to enter the food chain where it may lead to human exposure. Marine organisms are markedly enriched in plutonium compared to seawater. Concentration factors, defined as the ratio of the plutonium activity per unit mass of the fresh (wet weight) organism compared to the concentration in the same unit mass of seawater, for marine invertebrates in the North Atlantic Ocean range from 1 to 2 x 10^4. One of the most efficient collectors of plutonium is the seaweed *Sargassum*, and the standing crop of Pu-139, 240 in Sargasso weed in the Sargasso Sea is around 7.4 x 10^9 Bq.

Comparatively high plutonium concentrations are found in organisms such as marine worms which feed on sedimentary particles because surface sediments generally have higher plutonium concentrations than overlying waters. The blue mussel, *Mytilus edulis*, is a particularly efficient scavenger of plutonium, and has been extensively used as a biological indicator for radionuclides and trace metals[42].

Releases of Pu-238 caused by the stratospheric burn-up of the SNAP-9A reactor resulted in a tripling of the global inventory of this radionuclide, which is produced only in comparatively small quantities in nuclear weapons tests. Since 1964, SNAP-9A derived Pu-238 has been a valuable tracer for studying both the exchange of stratospheric air masses between the northern and southern hemispheres and the residence time of particles in the upper atmosphere. In Canadian waters, measurements of Pu-238 have been used to establish time stratigraphic horizons in the estuarine sediments of the Saguenay Fjord, Quebec[15]. The fact that the burn-up of a single SNAP-9A reactor in the upper atmosphere has resulted in the dissemination of a radioactive tracer measurable throughout the earth's lithosphere dramatizes the sensitivity of the environment to man-induced chemical perturbations.

There have been comparatively few measurements of americium or other transuranic radionuclides in the water column or suspended sediments. The distribution coefficients (K_d) for americium and curium are of the order of 2 x 10^6, but that for curium may be approximately 25% lower than that for americium[43]. Livingston and Bowen[44] have shown that in the Atlantic Ocean there is a significant increase in the Am-241 / Pu-239, 240 ratio below 1000 m in the water column. These results, together with the elevated values for this ratio found in freshwater systems, suggest that americium may be preferentially associated with particulate material. This conclusion is supported by the results of Koide *et al*[45] showing Am-241 / Pu-239, 240 ratios ranging from 0.5 to 3.2 in particulate phases of water samples collected in California coastal regions.

Livingston and Bowen[44] noted that the seaweeds *Fucus* and *Sargassum* discriminated against americium, while several other seaweeds and some invertebrates exhibited no preferential accumulation of any element. The U.S. Mussel Watch investigation analyzed plutonium and americium isotopes in the soft tissues of bivalves collected around the U.S. coastline[46]. Ratios of Am-241 / Pu-239 measured in the mussel, *Mytilus edulis*, from the east coast were around 2.5-3.1 while ratios in a different species of mussel, *Mytilus californianus*, collected on the west coast were around 0.2-0.3. The question to be answered is whether these results reflect differences in the bioconcentrating abilities of these species or differences in the availability of americium and plutonium on the east and west coasts.

URANIUM AND THORIUM DECAY SERIES

Although not pollutants, members of the uranium and thorium decay series are valuable as marine geochemical tracers and they can be used to establish transport pathways and mechanisms governing the oceanic distribution of anthropogenic trace metals and radionuclides.

Recent measurements of thorium distributions in the deep ocean indicate that reversible exchange occurs between dissolved and particulate phases throughout the water column. The kinetics of the adsorption and desorption processes, estimated by measuring several thorium isotopes in the same sample, indicate that reaction times of this element with particles are short (of the order of a few months) compared to the residence time of suspended matter in the deep ocean (several years). This implies that particles suspended in the deep sea are close to equilibrium with respect to the exchange of metals at their surfaces[47,48]. Th-230, Th-232 and Pa-231 are isotopes which have found wide applications in the determination of sedimentation rates in pelagic environments. Measurements of the ratios, Th-230/Th-232 and Pa-231/Th-230 can be used to estimate sediment geochronologies over a range of the order of 4 x 10^5 years. Th-234 also has interesting applications in studies of sediment dynamics, and its measurement in surficial sediments has been used to estimate the rates of very rapid bioturbation phenomena which occur in biologically productive regimes[49].

Most of the investigations of radium in seawater have focused on the longest lived isotope, Ra-226. In most oceanic regions there is a marked increase in the Ra-226 concentration as the sea floor is approached, a gradient which is maintained by the release of Ra-226 from interstitial waters of the sediments into the overlying water mass[50]. This introduction of a radioactive tracer ($t_{½} = 1620$ yr) at the sea floor has been exploited to determine rates of water mass mixing and to determine the importance of particulate transfer mechanisms in the deep ocean[51].

Recent work has focused on studies of Ra-228, a radionuclide which is introduced into surface ocean waters by release from inshore shelf and slope sediments. It is valuable as a tracer for calculating vertical eddy diffusion coefficients and also in the study of horizontal transport[52]. Recent measurements of Ra-228/Ra-226 ratios in waters of the Norwegian, Labrador and Greenland Seas and Baffin Bay have been used to characterize rates of horizontal and vertical mixing in these regions[53]. Pronounced concentration gradients in radon-222 ($t_{1/2} = 3.6$ days), the short-lived daughter of Ra-226, are produced both at the sea floor and at the water surface by boundary exchange processes[54]. Measurements of the deficiency of Rn-222 in surface waters has led to estimates of coefficients of vertical eddy diffusion and the rate of exchange of gas across the ocean-atmosphere boundary.

Because of their suitable half-lives and their accurately measured rates of supply to the oceans, Pb-210 ($t_{1/2} = 22.3$ yr) and Po-210 ($t_{1/2} = 138$ days) have become increasingly important as marine geochemical tracers. Both radionuclides have short residence times in the biologically productive, surface layers of the ocean. Pb-210 and Po-210 occur predominantly in the dissolved phase and their residence time of approximately 0.5 yr in the surface layer of the ocean probably characterizes their transfer from dissolved to particulate phases[55]. These radionuclides appear to be removed rapidly by biological activity, but the exact mechanism has not yet been established. Dissolved and particulate profiles for these radionuclides from the Labrador Sea, off Newfoundland, are similar to those measured in other parts of the Atlantic Ocean[56]. In the deep water, dissolved Po-210 is systematically depleted and particulate Po-210 enriched, indicating rapid association with particles following production from its grandparent, Pb-210. Dissolved Pb-210 exhibits a steadily increasing departure from secular equilibrium with Ra-226 with increasing water depth towards the seabed. This deficiency of Pb-210, observed throughout the deep Atlantic Ocean[57], is believed to arise from adsorption onto particles sinking in the water column and a less well-defined boundary scavenging process which involves direct uptake of Pb-210 at the sediment surface. Removal of Pb-210 from the deep ocean occurs on time scales comparable to its half-life of 22 years.

Pb-210 is therefore an excellent particle tracer for determining sedimentation rates in nearshore environments. The decrease in Pb-210 with increasing sediment depth can be used to determine sediment geochronologies and to establish the timing of trace metal and pollutant inputs to coastal and estuarine sediments during the 19th and 20th centuries. For example, Smith and Loring[58] were able to establish the detailed source for Hg pollution in the Saguenay Fjord, Quebec, through the analyses of Hg in a suite of sediment cores, which were dated using Pb-210. This work showed that the major source of Hg pollution in the Saguenay region was a chlor-alkali plant which began operations in 1947/48, resulting in a distinct Hg threshold in the sediments at this date. The sedimentary Pb-210 distribution may also contain a well-resolved record of both landslide and storm events, and pulsed inputs of material transported during periods of high river discharge[15]. Measurements of the Pb-210 distribution in low sedimentation, continental slope regimes, off Newfoundland, have been used to determine rates of sediment mixing by bioturbation[16] and similar measurements in fine-grained sediments in the Bay of Fundy have been used to identify regions in which sediment resuspension occurs.

REFERENCES

1. Preston, R. 1972. Artificial radioactivity in freshwater and estuarine systems. Proc., Roy. Soc. (London), 180B: 421-436.

2. Carter, M.W. and A.A. Moghissi. 1977. Three decades of nuclear testing. Health Phys., 33: 55-71.

3. Harley, N., I. Fissene, L.D. Ong and J.H. Harley. 1965. Fission yield and fission product decay. U.S.A.E.C. Health Safety Lab. Fallout Program, Quart. Summer Rep. Hasl - 164: 251-260.

4. Bowen, V.T., V.E. Noshkin, H.L. Volchok, H.D. Livingston and K.M. Wong. 1974. Cesium 137 to strontium 90 ratios in the Atlantic Ocean, 1966 through 1972. Limnology and Oceanography, 19: 670-681.

5. Bowen, V.T., V.E. Noshkin, H.D. Livingston and H.L. Volchok. 1980. Fallout radionuclides in the Pacific Ocean: vertical and horizontal distributions, largely from GEOSECS stations. Earth Planet. Sci. Lett., 49: 411-434.

6. Kupferman, S.L., H.D. Livingston and V.T. Bowen. 1979. A mass balance for Cs-137 and Sr-90 in the North Atlantic Ocean. J. Marine Res., 37: 157-199.

7. Smith, J.N., K. Ellis and F.J. Bishop. 1981. Pre-operational environmental monitoring report for the Point Lepreau, N.B., nuclear generating stations - 1980. Bedford Institute of Oceanography Report Series, BI-R-81-10, 111 p.

8. Alberts, J.J. and M.A. Wahlgren. 1981. Concentrations of Pu-239/240, Cs-137 and Sr-90 in the waters of the Laurentian Great Lakes. Comparison of 1973 and 1976 values. Environ. Sci. Tech., 15: 94-98.

9. Brunskill, G. Department of Fisheries and Oceans. Freshwater Institute, Winnipeg, Man. Personal communication.

10. Livingston, H.D., V.T. Bowen and S.L. Kupferman. 1982(a). Radionuclides from Windscale discharges I: nonequilibrium tracer experiments in high latitude oceanography. J. Mar. Res., 40: 253-272.

11. Livingston, H.D., V.T. Bowen and S.L. Kupferman. 1982(b). Radionuclides from Windscale discharges II: their dispersion in Scottish and Norwegian coastal circulation. J. Mar. Res. 40: 1227-1258.

12. Livingston, H.D., S.L. Kupferman, V.T. Bowen and R.M. Moore. 1984. Vertical profile of artificial radionuclide concentrations in the central Arctic Ocean. Geochim. Cosmochim. Acta, in press.

13. Smith, J.N., K. Ellis and F.J. Bishop. 1982. Pre-operational environmental monitoring report for the Point Lepreau, N.B., nuclear generating station - 1981. Can. Tech. Rep. Hydrog. Ocean Sci. No. 4, 200p.

14. Smith, J.N. and A. Walton. 1980. Sediment accumulation rates and geochronologies measured in the Saguenay Fjord using the Pb-210 dating method. Geochim. Cosmochim. Acta., 44: 225-240.

15. Smith, J.N. and K. Ellis. 1982. Transport mechanism for Pb-210, Cs-137 and Pu fallout radionuclides through fluvial-marine systems. Geochim. Cosmochim. Acta., 46: 941-954.

16. Smith, J.N. and C.T. Schafer. 1980. Bioturbation of surficial sediments on the continental slope, east of Newfoundland. Marine Radioecology (Proc., 3rd NEA Seina Seminar), NEA/OECD, Tokyo, 1979, 225-236.

17. Sugihara, T.T. and V.T. Bowen. 1962. Radioactive rare earths from fallout for study of particle movement in the sea. *In:* Radioisotopes in the Physical Sciences and Industry. International Atomic Energy Agency, Vienna.

18. Chesselet, R., D. Nordemann and C. Lalou. 1965. Etude comparative des taux de retombées radioactives maritimes et continentales. C.R. Acad. Sci., 260: 2875-2878.

19. Burton, J.D. 1966. Radioactive nuclides in the marine environment. *In:* Chemical Oceanography, 2nd Ed., Riley, Skarrow (Eds.), Vol. 3, 91-191.

20. Pentreath, R.J. 1977. Radionuclides in marine fish. Oceanogr. Mar. Biol. Ann. Rev., 15: 365-460.

21. Labeyrie, L.D., H.D. Livingston and V.T. Bowen. 1976. Comparison of the distributions in marine sediments of the fallout derived nuclides Fe-55 and Pu-239/240: a new approach to the chemistry of environmental radiouclides. *In:* Transuranium Nuclides in the Environment. IAEA, Vienna, 121 p.

22. Broecker, W.S. 1979. A revised estimate for the radiocarbon age of North Atlantic deep water. J. Geophys. Res., 84: 3218-3226.

23. Stuiver, M. 1980. C-14 distribution in the Atlantic Ocean. J. Geophys. Res., 85: 2711-2718.

24. Ostlund, H.G., H.G. Dorsey and C.G. Rooth. 1974. GEOSECS North Atlantic radiocarbon and tritium results. Earth Planet. Sci. Lett., 23: 69-86.

25. Jenkins, W.J. and P.B. Rhines. 1980. Tritium in the deep North Atlantic Ocean. Nature, 286: 877-880.

26. Brown, R.M. and W.E. Grummitt. 1956. The determination of tritium in natural waters. Can. J. Chem., 34: 220-226.

27. Dreisigacker, E. and W. Roether. 1978. Tritium and Sr-90 in North Atlantic surface water. Earth Planet. Sci. Lett., 38: 301-312.

28. Leventhal, J.S. and W.F. Libby. 1970. Tritium fallout in the Pacific United States. J. Geophys. Res., 75: 7628-7633.

29. Ostlund, H.G. and R.A. Fine. 1979. Oceanic distribution and transport of tritium. *In:* The Behaviour of Tritium in the Environment (International Atomic Energy Agency, Vienna, 1979), 315-336.

30. Weiss, W., W. Roether and E. Dreisigacker. 1979. Tritium in the North Atlantic Ocean: inventory, input and transfer into deep water. *In:* Behaviour of Tritium in the Environment. International Atomic Energy Agency, Vienna, 315-336.

31. Weiss, W. and W. Wolfgang Roether. 1980. The rates of tritium input to the world oceans. Earth Planet. Sci. Lett., 49: 436-446.

32. Mason, A.S. and H.G. Ostlund. 1979. Atmospheric HT and HTO: vertical distribution and large scale circulation, in behaviour of tritium in the environment. Proceedings of a Symposium, San Francisco, October 16-20, 1978, 3-16, IAEA, Vienna, 1979.

33. Hardy, E.P., P.W. Krey and H.L. Volchok. 1973. Global inventory and distribution of fallout plutonium. Nature, 241: 444-445.

34. Edgington, D.N. 1981. A review of the persistence of long-lived radionuclides in the marine environment - sediment/water interactions. *In:* Impacts of Radionuclide Releases into the Marine Environment. International Atomic Energy Agency, Vienna, 67-91.

35. Nelson, D.M. Argonne National Laboratory, Illinois. Personal communication.

36. Nelson, D.M. and M.B. Lovett. 1978. Oxidation state of plutonium in the Irish Sea. Nature, 276: 599-601.

37. Goldberg. E.D. 1954. Chemical scavengers of the sea. J. Geol., 62: 249 p.

38. Bowen, V.T., K.M. Wong and V.E. Noshkin. 1971. Plutonium-239 in and over the Atlantic Ocean. J. Mar. Res., 29: 1-10.

39. Miyake, Y. and Y. Sugimura. 1976. The plutonium content of Pacific Ocean waters. *In:* Transuranium Nuclides in the Environment. International Atomic Energy Agency, Vienna, 91-105.

40. Livingston, H.D. and V.T. Bowen. 1979. Pu and Cs-137 in coastal sediments. Earth Planet. Sci. Lett., 43: 29-45.

41. Noshkin, V.E. and K.M. Wong. 1980. Plutonium mobilization from sedimentary sources to solution in the marine environment. Marine Radioecology (Proc., 3rd NEA Seminar, Tokyo, 1979), NEA/OECD, Paris.

42. Goldberg, E.D. 1976. The Health of the Oceans. The Unesco Press, Paris.

43. Pentreath, R.J., D.F. Jefferies, M.B. Lovett and D.M. Nelson. 1980. The behaviour of transuranic and other long-lived radionuclides in the Irish Sea and its relevance to the deep sea disposal of radioactive wastes. Marine Radioecology (Proc., 3rd NEA Seminar, Tokyo, 1979), NEA/OECD, Paris.

44. Livingston, H.D. and V.T. Bowen. 1976. Americium in the marine environment-relationships to plutonium. Environmental Toxicity of Aquatic Radionuclides - Models and Mechanisms (Miller, M.W. and Stannard, J.N., Eds.). Ann Arbor Science.

45. Koide, M., P.W. Williams and E.D. Goldberg. 1981. Am-241/Pu-239 + 240 ratios in the marine environment. Mar. Environ. Res., 5: 241-246.

46. Goldberg. E.D., V.T. Bowen, J.W. Farrington, G. Harvey, J.H. Martin, P.L. Parker, R.W. Risebrough, W. Robertson, E. Schneider and E. Gamble. 1978. The mussel watch. Environ. Conserv., 5: 101-125.

47. Bacon, M.P. and R.F. Anderson. 1982. Distribution of thorium isotopes between dissolved and particulate forms in the deep sea. J. Geophys. Res., 87: 2045-2056.

48. Nozaki, Y., Y. Horibe and H. Tsubota. 1981. The water column distributions of thorium isotopes in the western North Pacific. Earth Planet. Sci. Lett., 54: 203-216.

49. Aller, R.C. and J.K. Cochran. 1976. Th-234/U-238 disequilibrium in nearshore sediment: particle reworking and diagenetic time scales. Earth Planet. Sci. Lett., 29: 37-50.

50. Chung, Y., H. Craig, T.L. Ku, J. Goddard and W.S. Broecker. 1974. Radium-226 measurements from three GEOSECS intercalibration stations. Earth Planet. Sci. Lett., 23: 116-124.

51. Broecker, W.S., Y.-H. Li and J. Cromwell. 1967. Radium-266 and Radon-222 concentrations in the Atlantic and the Pacific Oceans. Science, 158: 1307-1310.

52. Kaufman, A., R.M. Trier, W.S. Broecker and H.W. Feely. 1973. Distribution of Ra-228 in the world ocean. J. Geophys. Res., 78: 8827-8848.

53. Moore, W.S., H.W. Feely, and Yuan-Hui Li. 1980. Radium isotopes in sub-arctic waters. Earth Planet. Sci. Lett., 49: 329-340.

54. Broecker, W.S. 1965. An application of natural radon to problems in ocean circulation. Symposium on Diffusion in Oceans and Fresh Water. Lamont Geological Observatory, 1964, Palisades, N.Y., 116 p.

55. Nozaki, Y., J. Thomson and K. Turekian. 1976. The distribution of Pb-210 and Po-210 in the surface waters of the Pacific Ocean. Earth Planet. Sci. Lett., 32: 304-312.

56. Bacon, M.P., D.W. Spencer and P.G. Brewer. 1980. Lead-210 and polonium-210 as marine geochemical tracers: review and discussion of results from the Labrador Sea. *In:* Proceed. Sympos., Houston, Texas, April 23-28, 1978, 473-501.

57. Bacon, M.P., D.W. Spencer and P.G. Brewer. 1976. Pb-210/Ra-226 and Po-210/Pb-210 disequilibria in seawater and suspended particulate matter. Earth Planet. Sci. Lett., 32: 277-296.

58. Smith, J.N. and D.H. Loring. 1981. Geochronology for mercury pollution in the sediments of the Saguenay Fjord, Quebec. Environ. Sci. Tech., 15: 944-951.

INCIDENCE AND DISTRIBUTION OF PETROLEUM AND PETROLEUM PRODUCTS* IN THE WESTERN NORTH ATLANTIC

E.M. Levy**

No other pollutant can compete with petroleum and its refined products in terms of the amounts that enter the world ocean. Estimated at 6.1 million tonnes annually[1], the direct input from marine transportation, river runoff, offshore production, industrial and municipal wastes, etc., of "petroleum hydrocarbons" to the world ocean has resulted in a remarkably close relationship between the levels of surface pollution in the form of surface slicks and floating tar particles and the major tanker routes and shipping lanes[2]. Since less than 1% of the total world tanker traffic passes through Canadian waters between the Canada/United States border and Hudson Strait[3] and because the surface waters in this region are dominated by the southward flow of unpolluted Labrador Current water, surface contamination in this region is maintained at levels much below those farther south. For example, a sampling program carried out between 1971 and 1974 demonstrated that floating tar and fresh oil were almost nonexistent in Arctic regions and the Labrador Current except in areas immediately affected by discharges from ships or inputs from natural submarine seepage of petroleum[4,5]. Concentrations, however, were considerably higher in the Gulf Stream and values as high as 92 mg/m^2 were encountered in the Sargasso Sea[6]. During this period the overall level of tar pollution in the North Atlantic north of the Gulf Stream/North Atlantic Current system was virtually zero[4] while south of this the level was about 0.16 mg/m^2. A similar trend was shown by data collected at the U.S. Ocean Weather Stations in the western North Atlantic[7]. During April of 1981, tar concentrations across the Grand Banks[8] ranged from 0 at most stations to 85 µg/m^2 indicating that this portion of the Northwestern Atlantic is still in a pristine condition insofar as surface contamination by oil is concerned despite the activity of the oil exploration industry and several major tanker incidents off the east coast of Canada. There was evidence, however, that the former was having a detectable, albeit small and very localized, effect on the region. No evidence was found of oil remaining from the Kurdistan spill in 1980 because of the effectiveness of local current systems in transporting surface oil from the region. The results of previous studies on the Scotian Shelf following the ARROW spill[9] suggested that the oil from the Kurdistan would "disappear" from the region within a year following the spill and this, indeed, proved to be the case[10]. In the Gulf of St. Lawrence, concentrations of floating oil ranged from 0 in most areas to over 10 mg/m^2 in an area immediately affected by leakage of oil from a sunken barge[11].

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* *Petrochemicals; that is, substances obtained from petroleum by chemical reactions other than cracking and hydrogenation as carried out in the conventional refining of crude oil, are not included in this discussion.*

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This study also indicated clandestine discharges of oily wastes from ships near the Strait of Belle Isle and the entrance to the St. Lawrence River.

In addition to discrete liquid and solid phases that remain floating on the surface of the sea, some components of oil, notably those containing aromatic structures, have appreciable solubility in seawater and thereby readily enter the water column. Near a recent input of oil to the marine environment, higher-than-background concentrations of benzene derivatives and polyaromatic compounds are present in the water column and there is little doubt concerning their source. In addition, however, there is a low background of such substances even in waters and marine sediments from areas far removed from any direct anthropogenic input of oil, and it is generally thought that this results largely from fallout from the atmosphere of aromatic substances, including a wide variety of polynuclear aromatic hydrocarbons that are produced during the high temperature combustion of organic matter. Indeed, this fallout is so widespread that there scarcely remains an area, even in remote arctic regions, where these compounds cannot be detected by sensitive analytical procedures.

Since many of these compounds are toxic or have otherwise adverse effects on marine life, their total concentration provides a useful indication of marine environmental quality and an estimate of the upper limit for the degree of contamination from oil. Fortunately, this aggregate is readily measured by a simple but highly sensitive analytical procedure employing ultraviolet fluorescence spectrophotometry[12]. This makes it possible to analyze the large numbers of samples required to measure background levels on a broad geographical basis and to identify "hot spots" resulting from recent localized inputs such as oil spills or natural seepage of petroleum from the seabed. Although it is not possible to distinguish between the petroleum-derived and the other components of this background without resorting to much more sophisticated and costly methods of sampling and analysis, this procedure enables the biologically and environmentally important aromatic substances to be quantified. Accordingly, this approach is preferable to procedures based on analyses of aliphatic hydrocarbons which are produced in abundance in the marine environment and readily degraded by environmental processes. On this basis, a program has been conducted in this laboratory over the past several years to measure background levels of "dissolved/dispersed petroleum residues" in waters and surficial bottom off the east coast of Canada and the eastern Canadian Arctic (that is, the total concentration of the non-polar aromatic compounds expressed in terms of an equivalent amount of Bunker C fuel oil* from the tanker, ARROW).

In all areas studied, the frequency distribution histograms were highly skewed with most of the values near the lower end of the range of concentrations and comparatively few at the higher. Such distributions are best analyzed after logarithmic transformation of the data[13], and in most cases the transformed data passed the chi-square test for goodness of fit at the 95% confidence level. Thus, the geometric means are the appropriate measure of central tendency for these sets of data and as such provide a good estimate of the background level in the region sampled.* The background levels obtained for the Gulf of St. Lawrence, Scotian Shelf, Grand Banks and Labrador Sea are given in Tables 1-4.

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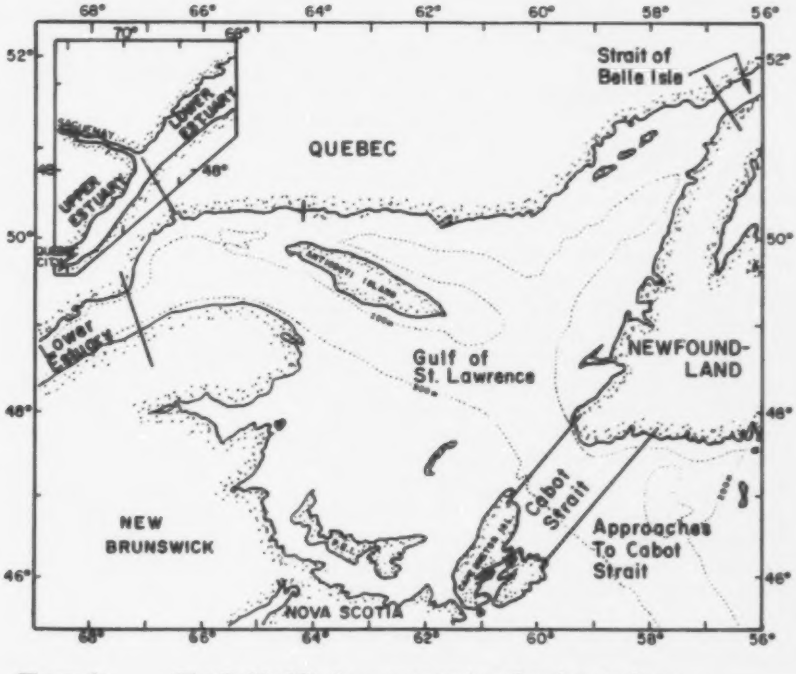
* *The intensity of fluorescence emission of this oil at 360 nm on excitation at 310 nm is 0.221 times that of chrysene under the same conditions.*

To interpret the distribution of dissolved/dispersed petroleum residues and any changes in the Gulf of St. Lawrence, the region was considered as 7 oceanographic entities (Fig. 1). In general, background levels (Table 1) were highest in the approaches to Cabot Strait, intermediate in Cabot Strait and lowest in the central region. This trend substantiates an earlier hypothesis based on preliminary studies of the area[11,14]; namely, that the major source of these substances in the Gulf of St. Lawrence is the Atlantic water that enters the Gulf through Cabot Strait and that their distribution within the Gulf is closely related to the subsequent movement of this water. Although the data bases are limited, there is no indication of a substantial input to the Gulf associated with the limited exchange of water through the Strait of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LOWER ESTUARY
QUEBEC CITY
Lower Estuary
QUEBEC
Strait of Belle Isle
ANTICOSTI ISLAND
200m
Gulf of St. Lawrence
NEWFOUNDLAND
Cabot Strait
Approaches To Cabot Strait
NEW BRUNSWICK
P.E.I.
NOVA SCOTIA

Figure 1: *The Gulf of St. Lawrence region showing study areas.*

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* *There was no relationship between concentration and depth in any of these sets of data.*

TABLE 1 - BACKGROUND LEVELS OF DISSOLVED/DISPERSED PETROLEUM RESIDUES IN THE GULF OF ST. LAWRENCE REGION, 1971-1979 (µg/L)

Cruise	Approaches to Cabot Strait	Cabot Strait	Gulf	Lower Estuary	Upper Estuary	Saguenay	Strait of Belle Isle	Dates
71027	6.3 (3)*	6.0 (21)	1.6 (200)	0.95 (18)	-	2.8 (15)	-	July/Aug, 1971
72017	3.0 (38)	4.6 (37)	2.9 (139)	-	2.7 (3)	3.7 (7)	2.3 (3)	May/June, 1972
73004	-	1.4 (33)	0.95 (78)	1.3 (7)	-	-	-	Feb/Mar, 1973
73012	2.3 (29)	1.6 (34)	1.1 (189)	1.9 (78)	-	1.5 (33)	-	Apr/May, 1973
74028	2.7 (141)	1.5 (44)	1.45 (238)	1.2 (23)	-	-	1.3 (31)	July/Aug, 1974
75015	-	0.40 (62)	0.42 (250)	0.43 (49)	-	-	-	May/June, 1975
76006	-	-	-	-	0.93 (85)	0.72 (64)	-	April, 1976
79024	-	0.62 (29)	0.35 (70)	0.35 (36)	1.1 (93)	-	-	Aug/Sept, 1979

* Numbers in parentheses are the number of values included in the geometric means.

Belle Isle. On the other hand, higher concentrations of dissolved/dispersed petroleum residues at the extreme upstream reaches of the upper estuary and Saguenay Fjord suggested there might be an appreciable local input of petroleum residues associated with river discharge. However, this is effectively removed from the system by processes associated with the mixing of fresh and saline waters and, therefore, does not constitute a major input to the Gulf region as a whole.

More intriguing, however, is that the data indicate a general decline in background levels throughout the region during the 70's. Presumably this is a consequence of the environmental movement of the late 60's and early 70's that led to legislation banning discharge of oily substances into the Gulf. Although such discharges still occur, pollution control measures, along with the associated surveillance and enforcement, seem to have resulted in a sufficient reduction in the input of oil to the Gulf and particularly to Cabot Strait and its approaches that an appreciable decrease in the background level of petroleum residues has occurred. Since the "residence time" of water in the Gulf is thought to be less than a year[15], any such reduction in the input would soon be reflected in lower background levels. It is not possible, however, from the data available to detect temporal changes in background levels on a shorter time scale; for example, possible seasonal effects related to spring runoff, winter ice conditions, etc. Although the most recent detailed survey of the Gulf was carried out in 1979, data collected in 1981 at one station in the Laurentian Channel south of Newfoundland indicated that the background level in the source water for the Gulf was 0.63 µg/L. This is in excellent agreement with the background measured in 1979 in Cabot Strait and, therefore, it would seem that the general improvement in environmental quality that occurred during the 70's has not been reversed.

Data collected during repeated studies along the Halifax section, a line of stations extending from the coast of Nova Scotia across the Scotian Shelf and into the deep water beyond also indicated a decline in background levels of dissolved/dispersed petroleum residues during the period of this program (Table 2). Following the KURDISTAN spill, concentrations of dissolved/dispersed petroleum residues both on the sea surface and in the water column increased dramatically[10] but, as in the case of the ARROW spill, concentrations returned to background within a few months after the input ceased.

Background levels in the waters of the Grand Banks in 1981 (Table 3) were the lowest anywhere in the region off the east coast of Canada. Slightly higher levels were observed in the southern portion of the Grand Banks than in the northern which is more strongly dominated by the Labrador Current and further removed from shipping. Somewhat higher levels were present in the Labrador Current in 1978 (Table 4) and still higher between Greenland and Flemish Cap in the Labrador Sea seaward of the Labrador Current.

TABLE 2 - BACKGROUND LEVELS OF DISSOLVED/DISPERSED PETROLEUM RESIDUES ON THE SCOTIAN SHELF, 1973-1975

Cruise	Background Level (µg/L)	Date
73002	2.4 (30)*	January, 1973
73012	7.3 (10)	April, 1973
74004	1.3 (43)	January, 1974
74032	3.0 (43)	September, 1974
75003	0.45 (33)	January, 1975
75026	0.35 (34)	October, 1975

TABLE 3 - BACKGROUND LEVELS OF DISSOLVED/DISPERSED PETROLEUM RESIDUES ON THE GRAND BANKS (Cruise 81008)

Region	Background Level (µg/L)	Date
Southern Banks	0.23 (61)*	April, 1981
Hibernia Area	0.16 (92)	April, 1981
South Tempest Area	0.13 (51)	April, 1981
Combined	0.17 (204)	April, 1981
Flemish Pass	0.20 (20)	April, 1981

TABLE 4 - BACKGROUND LEVELS OF DISSOLVED/DISPERSED PETROLEUM RESIDUES IN THE LABRADOR SEA (Cruise 78002)

Region	Background Level (µg/L)	Date
Western Labrador Sea	0.91 (130)*	Feb-Apr, 1978
Eastern Labrador Sea	0.76 (104)	Feb-Apr, 1978
Overall	0.84 (234)	Feb-Apr, 1978

* Numbers in parentheses are the number of values included in the geometric means.

It is evident from the data presented that there are small differences in background levels of petroleum at the sea surface and throughout the water column in the North Atlantic off the east coast of Canada. The concentration levels at any time and place depend upon the inputs and the oceanographic features. In most cases, the background contains substances from diffuse sources such as atmospheric fallout of combustion products but, in addition to this, there may be "point" inputs from anthropogenic sources. The recovery time; that is, the time for concentrations to return to normal after a major input, is about one year, or less. Most important, however, background levels throughout the region are well below those known to have adverse acute effects on most marine life although seabirds may not be able to tolerate any exposure to oil under conditions of high environmental stress. Similarly, concentrations seem to be below those that have been demonstrated to have low-level, long-term effects, although this subject is very poorly understood. Thus, insofar as contamination from petroleum residues is concerned, the entire region can be considered to be in a near pristine condition.

REFERENCES

1. Anon. 1975. Petroleum in the Marine Environment. National Academy of Sciences, Washington, D.C., 107 p.

2. Levy, E.M., M. Ehrhardt, D. Kohnke, E. Sobtchenko, T. Suzuoki and A. Tokuhiro. 1981. Global oil pollution: results of MAPMOPP, the IGOSS pilot project on marine pollution (petroleum) monitoring. Intergovernmental Oceanographic Commission, Paris, 35 p.

3. Anon. 1973. International Petroleum Encyclopedia. The Petroleum Publishing Co., Tulsa, Okla, 440 p.

4. Levy, E.M. and A. Walton. 1976. High seas oil pollution: particulate petroleum residues in the North Atlantic. J. Fish. Res. Board Can., 33: 2781-2791.

5. Levy, E.M. 1981. Background levels of petroleum residues in Baffin Bay and the eastern Canadian Arctic: role of natural seepage. *In:* Petroleum and the Marine Environment, Graham and Trotman Ltd., p. 345-362.

6. Levy, E.M. and J.D. Moffatt. 1975. Floating petroleum residue concentrations in the North Atlantic, 1971-1974. Bedford Institute of Oceanography Data Series, BI-D-75-9, December, 1975.

7. McGowan, W.E., W.A. Saner and G.L. Hufford. 1974. Tar ball sampling in the western North Atlantic. N.B.S. Special Publication, 409: 83-84.

8. Levy, E.M. 1982. Baseline levels of volatile hydrocarbons and petroleum residues in the waters and sediments of the Grand Banks. Can. J. Fish Aquatic Sci. 40, Supplement No. 2,22-33

9. Levy, E.M. 1972. Evidence for the recovery of the waters off the east coast of Nova Scotia from the effects of a major oil spill. Water, Air, Soil Pollut., 1: 144-148.

10. Lawrence, D.J., E.M. Levy and MacLaren Plansearch Ltd. 1981. Directory for oceanographic data collected following the Kurdistan oil spill. BIO Data Series.

11. Levy, E.M. and A. Walton. 1973. Dispersed and particulate petroleum residues in the Gulf of St. Lawrence. J. Fish. Res. Board Can., 30: 261-267.

12. Levy, E.M. 1977. Fluorescence spectrophotometry: principles and practice as related to the determination of dissolved/dispersed petroleum residues in seawater. BIO Report Bi-R-77, July, 1977.

13. Aitchison, J. and J.A.C. Brown. 1969. The lognormal distribution. Cambridge University Press, London, 176 p.

14. Levy, E.M. 1971. The presence of petroleum residues off the east coast of Canada, in the Gulf of St. Lawrence and the St. Lawrence River. Water Res., 5: 723-733.

15. Trites, R. 1970. The Gulf of St. Lawrence from a pollution viewpoint. Unpublished Manuscript.

ORGANOHALOGEN COMPOUNDS

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The principal organohalogen compounds which I discuss here are the organochlorines (OCs), especially the DDT group of insecticides and the polychlorinated biphenyls (PCBs). Although other OC insecticides such as chlordane, toxaphene and mirex are now attracting attention, there are too few data describing their occurrence to allow any inference of spatial or temporal trends in their distribution.

Both DDT and PCBs are purely synthetic: they are manufactured exclusively by man, but in the environment they may be changed by natural processes such as photolysis or by enzymatic degradation. DDT was, and still is, produced for use as a broad spectrum insecticide: the technical product contains mainly *p,p'*-DDT (2,2-bis-(-chlorophenyl)-l,l,l,-trichloroethane) with about 20% of the *o,p'*-isomer. In the environment, *p,p'*-DDT is usually found with two principal breakdown products, *p,p'*-DDD and *p,p'*-DDT (2,2-bis-(*p*-chlorophenyl)-1,1-dichloroethane and 2,2-bis-(*p*-chlorophenyl)-l,l-dichloroethylene, respectively). *p,p'*-DDD is itself an insecticide, but *p,p'*-DDE seems to be a detoxification product. For convenience in this chapter, I shall refer to *p,p'*-DDT, its *o,p'*-isomer, and their metabolites collectively as ΣDDT. More details about the chemistry of *p,p'*-DDT can be found in various reference texts (e.g., O'Brien[1]). The main use of DDT in Eastern Canada was probably in the New Brunswick forest spraying programme: from the early 1950's up to the early 1960's, technical DDT was sprayed in amounts up to a maximum (in 1957) of about 6×10^6 kg, annually[2]. This figure is equivalent to about 10% of the total U.S. production of DDT in 1957 (56.6×10^3 metric tons[3]). Some of the ΣDDT found in the North West Atlantic ecosystem undoubtedly comes from this source, either by atmospheric transport or by river drainage. Of course, both processes could also transport ΣDDT from other areas where it was used, such as the Great Lakes Basin.

PCBs have had only a small scale and local industrial application in Eastern Canada[4]; on the other hand, they were components of a variety of consumer products which were widely used (hydraulic fluids, pressure sensitive copying paper) and whose disposal was not controlled. Because of this, and because also they are easily transported in the atmosphere from other areas where they might be more widely used, they seem to be distributed fairly generally throughout the marine environment of Eastern Canada.

In this chapter, I try to assess the state of contamination of the North West Atlantic ecosystem by OC compounds. I do so by comparing the OC residue concentrations reported for samples taken from this area with those taken elsewhere, and I also try to relate the levels of OCs found in local samples to potential deleterious effects. However, it is worthwhile ex-

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plaining why the data available may not allow us to make conclusive statements about these subjects.

Surveys of the OC levels in the Northwest Atlantic ecosystem have been carried out only since the late 1960's. Initially, these analyses were carried out on fairly random samples of marine organisms or products, with the aim of ascertaining what contaminants were present and in what amounts. Since the emphasis was on marine products for human consumption, sampling was concentrated in the productive fishing zones, and tended to focus on species of commercial importance. It was not until the early 1970's that any attempts were made to assess trends or differences in the level of contamination by measuring OC levels in various environmental samples, including marine biota: by this stage, it was recognized that various physical and biological processes, as well as the ambient contamination levels, would affect OC concentrations in marine biota, but these sources of variation were rarely eliminated in the early surveys. A final source of difficulty in assessing the significance of any OC residue determination is that different laboratories vary in the accuracy with which they can carry out OC analyses (e.g. Holden and Topping[5]). The outcome of all this is that it is extremely difficult to detect confidently any subtle differences in ambient OC levels in different places or at different times. We can therefore make only rather general statements about the level of OC contamination in the North West Atlantic, and we can make only vague predictions of how this will vary during the next few years.

ORGANOCHLORINE CONCENTRATIONS IN THE NORTH WEST ATLANTIC

The Water Column

OC compounds are very insoluble in water. They are therefore readily absorbed to, and reluctantly desorbed from small particulate material in the sea[6,7]. Because OCs are generally lipid soluble, they also tend to concentrate at the lipid-rich surface microlayer, usually the top few hundred microns[8]. In principle, sampling and analysis of OCs should consider both these sources of bias, but in practice, only the latter is eliminated, usually by default: it is difficult to sample the surface microlayer.

Typical ΣDDT concentrations in surface waters (excluding the microlayer) range from about 0.1 to 10 ng./L; PCB concentrations in the same medium are about ten times higher - approx. 1 to 100 ng./L. Concentrations of both residues in the microlayer may be about 5 times higher again. Little information has been published describing OC concentrations in the area covered by this review, but elsewhere in the Western North Atlantic, both PCB and ΣDDT levels fall into the ranges listed above[9,10]. PCB concentrations in the waters of St. George's Bay, N.S., also fall within the range listed[11].

It is difficult to interpret these results, partly because of the shortage of comparable data and partly because we know relatively little of how OC concentrations in sea water vary under normal conditions. We expect that sea water OC concentrations may fluctuate rapidly, probably reflecting short term changes in the amounts and concentrations of atmospheric input[12]. Unless these sources of variation are eliminated, it is difficult to infer any spatial or temporal trends in the distribution of OCs in the water column.

Marine Sediments

Relatively few data are available describing sediment OC concentrations in Eastern Canada. In general, marine sediments which receive only "background" amounts of contamination can be expected to contain up to about 50 ng/g each of ΣDDT and PCBs, and about 10 ng/g of cyclodiene insecticides. Concentrations over about 100 ng/g of ΣDDT or of PCBs probably indicate some nearby source of contamination such as sewage or an industrial outfall and increasing OC concentrations are usually associated with increasing organic carbon levels[13]. The only published data relevant to the area of study are those of Cannone and Mamarbachi[14] who found St. Lawrence estuary sediments to contain ΣDDT ranging from trace amounts up to 34 ng/g. Harvey and Steinhauer[15] found OC concentrations in open ocean sediments from the southern North West Atlantic to range from 0.1 to 3 ng/g. Thus, sediments from this area probably contain OC concentrations within the general range of values expected for "background" contamination.

Invertebrates

It has become clear during the last few years that OC residue concentrations in marine (and other) biota are affected by several biological factors such as age, sex and nutritional or reproductive status, as well as by the extent of ambient environmental contamination. For technical reasons, it is often difficult to eliminate these factors in studies of residue levels in invertebrates (even if their influence is recognised) and so comparison of samples differing in space or time is difficult.

Residues of various OCs have been reported in various samples of bivalve molluscs. ΣDDT levels vary around 0.02 µg/g wet weight[16,17,18,19]. PCB concentrations in molluscs have been reported only by Sims *et al.*[19]; values vary around 0.02 µg/g except for one small sample of oysters (0.003 µg/g. These values are at the lower end of the range of values reported during the U.S. National Shellfish Monitoring Survey[20], and are lower than some of the values reported from more obviously contaminated industrial areas (summarised by Pearce et al.,[17]). However, OC residue levels in molluscs are likely to be influenced by so many factors other than ambient contaminant levels that it is difficult to infer any trends from the data.

Residues of OCs have been reported in euphausiid shrimp from the Gulf of St. Lawrence[21]. Levels ranged from approx. 10 to 60 µg/g wet weight, and there were significant differences between the genera studied *(Meganyctiphanes* and *Thysanoessa);* this may have reflected trophic differences, or differences in lipid metabolism. No spatial differences between the outer Gulf of St. Lawrence and Eastern Cape Breton Island were seen. Concentrations were within the ranges described for open ocean Atlantic zooplankton[22].

Finfish

Both experiment and observation have shown that OC levels in fish vary with age (reflecting length of exposure: e.g., Bache *et al.*[23]), sex and reproductive status. The latter variables affect lipid metabolism and mobilisation, and hence the retention of OCs (e.g., Guiney *et al.*[24]). These variables have been eliminated from the more recent sampling programmes, especially those undertaken to compare spatial or temporal difference in contamination of fish stocks. As a result, such differences or trends may be more reliably detected by monitoring fish than by monitoring samples from lower trophic levels.

Residues of either the PCBs or the ΣDDT group vary from about 0.01 to approx. 5.0 μg/g wet wt., depending on the variables listed above, the time and place of sampling, the tissues analysed, and of course, on the species and trophic level. This range is too large to be useful for comparisons over space or time, and so monitoring programs have focussed on a more closely defined sample. Among the most comprehensive of these so far has been the ICES Baseline Monitoring Study[25] of 1975 in which defined species of fish, selected for year class and season, were compared. Although there were difficulties with the study, some general conclusions emerged. Cod liver (the tissue most often examined) contained a range of ΣDDT and PCB levels, as well as various other OCs including hexachlorocyclohexanes and dieldrin. Samples from Eastern Canada (Gulf of St. Lawrence and Scotian Shelf) contained ΣDDT levels towards the high end of the range; these concentrations were similar to those found in the North Sea, and higher than levels found in Greenland or Spitzbergen. PCB levels in Eastern Canadian fish were intermediate, being higher than those found in Arctic samples, but lower than those found in, say, the southern North Sea, where there are major industrial inputs. Dieldrin levels in Eastern Canadian fish samples also tended to be relatively high.

The general conclusion that ΣDDT contamination of local marine fish was significant in the early 1970's (and still may be) is borne out by comparisons of analyses of local herring with those of Jensen *et al.*[26] in the Baltic. The Swedish study was undertaken to assess spatial variations in ΣDDT concentrations by sampling commercial herring catches; some control over the size and season of catch was exercised. Various analyses of Canadian herring samples[19,27] show that typical ΣDDT levels are usually below 0.5 μg/g wet wt.; this is intermediate in the range of levels found in the Swedish sampling (0.2 μg/g for uncontaminated areas, to 1.7 μg/g for more highly contaminated areas).

Seals

As is the case in fish, the variation of OC residue concentrations with age and sex[28,29] is now reasonably well established. It is possible to eliminate these variations as sources of error in making comparisons of ambient contaminant levels by examining seal residues.

OC levels in grey seals *(Halichoerus grypus)* are lower in Eastern Canada (Sable Is.) than in the North Sea. Although ΣDDT levels are approximately similar[30,31,32], PCB concentrations in the Sable Is. samples are about half those in the Farne Is., and well below those in East Anglia. However, OC residues in grey and harp *(Pagophilus groenlandicus)* seals from several points in Eastern Canada[28,30,33] are about one order of magnitude higher than those in the ringed seal *(Pusa hispida)* in the Canadian Western Artic — a relatively uncontaminated area[29]. (Although the species compared here differ, they occupy roughly similar ecological niches and would be expected to be exposed to roughly similar OC sources if ambient levels were similar). Direct comparisons between Canadian Arctic ringed seals and ringed seals from the highly contaminated Gulf of Bothnia show that the latter contain ΣDDT and PCB concentrations about 100 times higher[34]. Taking all the seal data together, it seems that OC residues in Eastern Canada are higher than those from the uncontaminated Canadian Arctic, but appreciably lower than those from the highly contaminated Baltic.

Conclusions

Residue levels in organisms in Eastern Canada from lower trophic levels are similar to those found elsewhere; factors other than ambient concentrations which control levels in these biota are unknown or are uncontrolled, so that spatial trends cannot be inferred from these analyses. OC residues in finfish from Eastern Canada cover a wide range; when factors other than ambient levels are eliminated, it becomes clear that this region is relatively contaminated by ΣDDT, and less so by PCBs, in comparison with areas in Western Europe. Residue data from seals confirm this tentative conclusion, and show that Eastern Canada is more contaminated than the relatively pristine Arctic, and less contaminated than the obviously polluted Baltic.

TEMPORAL TRENDS

Short-term fluctuations in the input of OCs from the atmosphere to the sea are likely to be reflected first in low trophic level organisms such as phytoplankton[12]. We could predict that a permanent reduction in the presence of PCBs in the atmosphere - and hence in input to the sea - such as that resulting from the reduction in PCB use and manufacture in the early 1970's would also be detected first in plankton. No formal monitoring programme to record this possible trend has been established, but analyses of PCB concentrations in plankton from the Southern Gulf of St. Lawrence between 1973 and 1976 show a reduction of about 1,000 fold in PCB levels over this period (G.C. Harding, unpublished data). Furthermore, the high correlation observed in 1973 between plankton PCB levels and cumulative rainfall (suggesting atmospheric input) was not found in the 1976 analyses; this suggests that the source of PCBs to the plankton differed. Unfortunately, differences between the sampling and analytical procedures in the two series of analyses preclude rigorous comparisons, but the data suggest a declining trend in PCB concentrations in the area.

At higher trophic levels, the response to even sudden change in OC input to the North Atlantic ecosystem might be more gradual. (This is borne out by the U.S. National Shellfish Monitoring Programme, which showed a gradual rise to a maximum in the late 1960's and then a gradual decline; in this case, riverine input of several OCs was being recorded.) However Zitko[35] has summarised some data which suggests that there may be a gradual and slight decline in OC contamination of Gulf of St. Lawrence and S.W. Nova Scotia herring stocks during the early 1970's. A clearer picture of this trend (if it exists) will have to wait for several more years' worth of analysis.

Comparisons of organochlorine concentrations in Eastern Canadian seals sampled during the early to mid 1970's and 1982 show some declining trends during the sampling intervals. In female grey seals *(Halichoerus grypus)* from Sable Is., ΣDDT concentrations fell by about half between the mid 1970's and 1982, but PCB concentrations did not change. In male harp seals *(Pagophilus groenlandicus)* from the Gulf of St. Lawrence, ΣDDT concentrations fell about 4-fold between 1971 and 1982; PCBs appear to have fallen marginally (less than half) during the same period (R.F. Addison, unpublished data).

SIGNIFICANCE OF PRESENT OC LEVELS

The OC levels which are currently found in water or marine biota in Atlantic Canada are well below those which are associated with acute effects. Thus, short-term concentrations of around 1 ng/L or higher are needed to kill aquatic biota; these concentrations are at least 100 and more often 1000 times higher than ambient water concentrations. It is unlikely that our beaches will be littered with fish killed by OC poisoning. However, less acute effects can be detected: these include changes in the normal metabolism of hormonal steroids, and the induction of certain enzymes. These effects can be brought about by levels of some contaminants — especially PCBs — now found in local marine fish. However, although sublethal effects can be detected at the organisation level of enzymes or biochemical processes, it is difficult to show that they have any effect at the whole-organism, population or community level. Only in the case of the Baltic ringed seals, which contain PCBs in the range of several hundred μg/g (more than ten times higher than local levels) is there even a suggestion that population viability may be affected. In that case, the high PCB concentrations are believed to influence reproductive success.

A final word concerns the public health aspect of contaminant levels in marine biota. Most OC levels in Atlantic Canadian fish and fishery products are well below the current tolerances imposed by the regulatory agencies (2 μg/g for PCBs, 5 μg/g for ΣDDT; both as wet wt. of edible portion). Only in the cases of very large or relatively old fish (swordfish and tuna) does it appear likely that these tolerances would be exceeded. In most other cases, levels are well within the tolerances, or else tissues with high OC levels are discarded or are processed to remove OC contaminants[36].

REFERENCES

1. O'Brien, R.D. (1967). Insecticides: Action and Metabolism. Academic Press, N.Y.

2. Kerswill, C.J. (1967). Studies on effects of forest spraying with insecticides, 1952-63, on fish and aquatic invertebrates in New Brunswick streams: Introduction and summary. J. Fish. Res. Bd. Canada 24: 701-708.

3. Anon. (1970). Man's impact on the global environment. Report of the Study of Critical Environmental Problems, M.I.T. Press, 1970.

4. Anon. (1977). PCBs in the Atlantic Provinces. Surveillance Report EPS-5-AR-77-13 EPS, Atlantic Region, 1977.

5. Holden, A.V. and Topping, G. (1981). Report on further intercalibration analyses in ICES pollution monitoring and baseline studies. ICES Cooperative Research Report No. 108, Copenhagen, 1981.

6. Pierce, R.H., Olney, C.E. and Felbeck, G.T. (1974). *p,p'*-DDT adsorption to suspended particulate matter in seawater. Geochim. Cosmochim. Acta. 38: 1061-1073.

7. Hargrave, B.T. and Kranck, K. (1976). Adsorption and transport of pollutants on suspended particles. Proc. Symp. Non-biological. Transport of Pollutants, Gaithersburg, Md.

8. Seba, D.B. and Corcoran, E.F. (1969). Surface slicks as concentrators of pesticides in the marine environment. Pest. Monit. J. 3: 190-193.

9. Harvey, G.R., Steinhauer, W.G. and Teal, J.M. (1973). Polychlorinated biphenyls in North Atlantic Ocean water. Science 180: 643-644.

10. Jonas, R.B. and Pfaender, P.K. (1976). Chlorinated hydrocarbon pesticides in western North Atlantic Ocean. Env. Sci. Technol. 10: 770-773.

11. Stadler, D., German Hydrographic Institute, Kiel. Personal Communication.

12. Ware, D.M. and Addison, R.F. (1973). PCB residues in plankton from the Gulf of St. Lawrence. Nature 246: 519-521.

13. Addison, R.F. (1980). A review of organohalogens with respect to the Ocean Dumping Control Act, pp 13-21. *In* Regulated levels of scheduled I substances in the Ocean Dumping Control Act - A review. J.J. Swiss, R.F. Addison, D.W. McLeese and J.F. Payne. Ocean Dumping Report 3, Dept. Fisheries and Oceans, Ottawa, 1980.

14. Cannone, P. and Mamarbachi, G. (1975). Résidus des insecticides organochlorés dans les sediments du Haut Estuaire du Fleuve St. Laurent. Bull. Env. Contam. Toxicol. 14: 83-87.

15. Harvey, G.R. and Steinhauer, W.G. (1976). Biogeochemistry of PCB and DDT in the North Atlantic. *In* Environmental Biogeochemistry, Ann Arbor Sci. Publ., Ann Arbor, Mich. pp 203-221.

16. Sprague, J.B. and Duffy, J.R. (1971). DDT residues in Canadian Atlantic fishes and shellfishes in 1967. J. Fish. Res. Bd. Canada 28: 59-64.

17. Pearce, P.A., Gruchy, M. and Keith, J.A. (1973). Toxic chemicals in living things in the Gulf of St. Lawrence. Canadian Wildlife Service Pesticide Section Manuscript Report 24, Ottawa, 1973.

18. Hargrave, B.T. and Phillips, G.A. (1976). DDT residues in benthic invertebrates and demersal fish in St. Margaret's Bay, N.S., J. Fish. Res. Bd. Canada 33: 1692-1698.

19. Sims, G.G., Campbell, J.R., Zemlyak, F.L., Graham, J.M. (1977). Organochlorine residues in fish and fishery products from the Northwest Atlantic. Bull. Env. Contam. Toxicol. 18: 697-705.

20. Butler, P.A. (1973). Organochlorine residues in estuarine mollusks, 1965-72-National Pesticide Monitoring Program. Pest. Monit. J. 6: 238-362.

21. Sameoto, D.D., Darrow, D.C. and Guildford, S. (1975). DDT residues in euphausiids in the Upper Estuary of the Gulf of St. Lawrence. J. Fish. Res. Bd. Canada 32: 310-314.

22. Riseborough, R.W., Vreeland, V., Harvey, G.R., Miklas, H.P. and Carmignani, G.M. (1972). PCB residues in Atlantic zooplankton. Bull. Env. Contam. Toxicol. 8: 345-355.

23. Bache, C.A., Serum, J.W., Youngs, W.D. and Lisk, D.J. (1972). Polychlorinated biphenyl residues: accumulation in Cayuga Lake trout with age. Science. 177: 1191-1192.

24. Guiney, P.D., Melancon, M.J., Lech, J.J. and Peterson, R.E. (1979). Effects of egg and sperm maturation and spawning on the distribution and elimination of a polychlorinated biphenyl) in rainbow trout *(Salmo gairdneri)*. Tox. Appl. Pharmacol. 47: 260-272.

25. ICES (1977). A baseline study of the level of contaminating substances in living resources of the North Atlantic. ICES Cooperative Research Report No. 69, Copenhagen.

26. Jensen, S., Johnels, A.G., Olsson, M. and Otterlind, G. (1972). DDT and PCBs in herring and cod from the Baltic. Ambio Spec. Rept. 1: 71-85.

27. Zitko, V. (1971). Polychlorinated biphenyls and organochlorine pesticides in some freshwater and marine fishes. Bull. Env. Contam. Toxicol. 6: 464-470.

28. Addison, R.F., Kerr, S.R., Dale, J. and Sergeant, D.E. (1973). Variation of organochlorine residue levels with age in Gulf of St. Lawrence harp seals *(Pagophilus groenlandicus)*. J. Fish. Res. Bd. Canada 30: 595-600.

29. Addison, R.F. and Smith, T.G. (1974). Organochlorine residue levels in Arctic ringed seals: variation with age and sex. Oikos 25: 335-337.

30. Addison, R.F. and Brodie, P.F. (1977). Organochlorine residues in maternal blubber, milk and pup blubber from Grey seals *(Halichoerus grypus)* from Sable Is., Nova Scotia. J. Fish. Res. Bd. Canada 34: 937-941.

31. Heppleston, P.B. (1973). Organochlorines in British grey seals. Mar. Poll. Bull. 4: 44-45.

32. Donkin, P., Mann, S.V. and Hamilton, E.I. (1981) Polychlorinated biphenyl, DDT and dieldrin residues in grey seals *(Halichoerus grypus)* males, females and mother-foetus pairs sampled at the Farne Is., England during the breeding season. Sci. Total Env. 19: 121-142.

33. Frank, R., Ronald, K. and Braun, H.E. (1973). Organochlorine residues in harp seals *(Pagophilus groenlandicus)* caught in Eastern Canadian waters. J. Fish. Res. Bd. Canada 30: 1053-1063.

34. Helle, E., Olsson, M. and Jensen, S. (1976). DDT and PCB levels and reproduction in ringed seals from the Bothnian Bay. Ambio 5: 188-189.

35. Zitko, V. (1981). Monitoring program for major Atlantic coast fisheries. Can. Manuscript Rept. Fish. Aq. Sci. No. 1615.

36. Addison, R.F. and Ackman, R.G. (1974). Removal of organochlorine pesticides and polychlorinated biphenyls from marine oils during refining and hydrogenation for edible use. J. Amer. Chem. Soc. 51: 192-194.

NON-HALOGENATED SYTHETIC ORGANIC CHEMICALS

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INTRODUCTION

The bulk of data on specific chemical species in marine matrices deals with organochlorine materials, in particular DDTs and PCB's. In large measure, this is attributable to the environmental problems associated with the widespread use of such materials for agricultural and industrial purposes, both in North America and Europe. The demonstrated widespread environmental occurrence of these materials has led, in turn, to the development of sophisticated analytical tools (e.g., the electron capture gas chromatographic detector) highly specific to their detection.

The potential environmental and health hazards ascribed to many halogenated organic compounds has resulted in a marked decline in the use of this class of chemicals over the past 10-15 years, and a subsequent increase in the use of non-halogenated materials as substitutes. Consequently, there has been a gradual shift in the emphasis of environmental studies towards determining the environmental behaviour and effects of non-halogenated organic chemicals. Unfortunately, it will be some time before detailed data bases are established which will permit assessment of spatial and temporal trends in distribution within particular environmental compartments. Two factors which have restricted the acquisition of such information are the lack of consensus as to which of these new chemicals pose the greatest concern, and the high cost of analytical instrumentation which will afford the necessary specificity and sensitivity.

A number of entry routes for organic compounds into the marine environment exist, the most important being: (1) direct discharge, (2) atmospheric transport resulting from volatilization and wind erosion, (3) ground water runoff, and (4) movement in suspension of a soil-chemical complex[1]. All of these will have some bearing on the situation in the local marine environment.

As already mentioned, information on non-halogenated synthetic organic chemicals is sparse, both in regard to the Northwest Atlantic and other marine ecosystems which could be used for comparison and inference. The data available permit only a cursory overview.

PESTICIDES

In Atlantic Canada substantial quantities of non-halogenated biocides are used for pest control. Biocides employed in the New Brunswick forest spray program are applied over a considerable geographical area; while for more localized problems, such as the control of agricultural pests and industrial applications, the use is intensive but target areas are relatively limited in size.

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Agricultural Application

Some biocides commonly used for agricultural purposes in the region are mancozeb, disulfoton, paraquat, maneb, zineb, phorate, aldicarb, dinoseb, carbaryl and carbofuran[2]. Unfortunately, little enviromental monitoring for these materials has been conducted to date.

A limited survey[3] for maneb (the most intensively applied biocide in the region) in estuarine sediments did not detect any parent compound at a minimum detectable level of 0.2 µg/g.

Alkyldinitrophenols, which include such materials as dinoseb and dinocap, are often used as fungicides and herbicides. In spite of the relatively high dosage rates of some of these compounds (3.6-6.8 kg/ha of dinoseb for potato-vine killing), little attention has been paid to their effects on the aquatic environment. Dinoseb has been reported[4] to have 24- and 96-h LC50's of 0.18 and 0.11 mg/L, respectively to speckled trout *(Salvelinus fontinalis)*. This pesticide has been detected in a New Brunswick stream[5] at a concentration of 4.81 mg/L.

Zitko[5] assessed the acute toxicity of eighteen 2-alkyl-4,6-dinitrophenols. Certain isomers were found to be extremely lethal to juvenile Atlantic salmon *(Salmo salar)*, larval and adult lobsters *(Homarus americanus)*, but comparatively non-toxic to crayfish *(Oronectes limosus)*. The toxicity of the alkyldinitrophenols tested increased with increasing octanol/water partition coefficients. He concluded, however, that these compounds were unlikely to bioaccumulate in the aquatic environment and residue levels would not be detected in aquatic fauna.

Carbaryl has been shown[6] to induce cardiovascular abnormalities and developmental arrest in salt water killifish *(Fundulus heteroclitus)* at levels approximating 1-10 mg/L. In addition, optic abnormalities in the Atlantic silver-side *(Menidia menidia)* at carbaryl concentrations down to 0.01 mg/L in the water column were reported.

In general, organophosphorus, carbamate and alkyldinitrophenol biocides tend to be short lived in the abiotic environment and biological systems. Organophosphorus compounds, for instance, degrade rapidly through hydrolysis and other reactions to non-toxic, water-soluble compounds which are rapidly excreted by aquatic animals[7].

Agricultural pesticides may, through improper application or exposure, pose localized estuarine problems; however, little danger is represented by these chemicals to the offshore ecosystem since lethal and sublethal thresholds for marine species tested so far are considerably in excess of anticipated environmental levels.

Forest Spraying

Two biocides, fenitrothion and aminocarb, are of particular interest in the Atlantic Region. These two compounds, in differing spray formulations, have been used since 1969 as the principal control agents for the spruce budworm in New Brunswick. In total, some 5,500,000 kg of fenitrothion and 600,000 kg of aminocarb have been aerially applied.

Aminocarb has been found to have few adverse short term effects on freshwater non-target organisms over a variety of application rates[8]. In the freshwater environment, adult crayfish exhibited symptoms of acute toxicity at concentrations of 10 ppm or less, with mortality occurring above 25 ppm[9]. The bioaccumulation and excretion rates of carbamate pesticides, such as aminocarb, by aquatic species[10] indicate that fish and aquatic invertebrates do not concentrate these compounds to any great degree. Based on the low accumulation coefficients determined for mussels *(Mytilus edulis)*, it appears that aminocarb concentrations in water of less than 0.01 mg/L are not likely to result in significant contamination of bivalves[11]. Consequently, it is not surprising that no aminocarb residues were detected in shellfish collected from New Brunswick estuaries 1-14 days post spray[12].

An aspect of the aerial application of aminocarb that until recently has received little attention is the fate and effects of other components of the spray formulation. Two such components that have recently been tested are nonylphenol and pesticide diluent 585. Laboratory studies have indicated that nonylphenol is not, as it is often labelled, an "inert" ingredient. For instance, for juvenile Atlantic salmon the lethality of the aminocarb forumulation is due almost entirely to nonylphenol[13]. Among the animals tested in these studies, the order of sensitivity to formulation compounds appeared to be: marine crustaceans greater than salmon greater than bivalves.

Nonylphenol was used as a spray component between 1978 and 1980 during which time roughly 1,100,000 kg were deposited on New Brunswick forests. Environmental monitoring for nonylphenol in the region has been limited and no data pertaining to the marine environment are available. Nonylphenol residue concentrations in the surface water of a New Brunswick lake were maximal (12.0 µg/L) at one hour post-spray, and undetectable (<1.0 µg/L) after 66 hours[14]. This finding confirms the supposition that nonylphenol will be short lived in the aquatic environment.

The organophosphate insecticide fenitrothion, while having a toxicity to fish much lower than DDT, can have adverse environmental effects. Forest spraying of fenitrothion causes little direct mortality of young salmon[15]; however, the insecticide may alter territorial behaviour and locomotion of the fish.

Shellfish collected in two estuary systems in New Brunswick contained fenitrothion residues[12,16], even when sample collection was several days after application. In this instance, fenitrothion apparently entered the estuarine environment and, by extension, the marine ecosystem. Additional post-spray monitoring in the same areas indicated rapid depuration of the insecticide from contaminated species. This finding was contradicted by a report[17] detailing laboratory studies which demonstrated that for softshelled clams *(Mya arenaria)*, mussels *(Mytilus edulis)* and fresh water clams *(Anodonta cataractae)* fenitrothion has a low uptake and a high excretion rate, combining for a low accumulation coefficient. Consequently, no danger of shellfish contamination as a result of normal application procedures is anticipated.

Fenitrothion is absorbed or actively taken up by plankton, particularly the phytoplankton[18]. The degree of absorption depends on population numbers and various environmental conditions, particularly seasonal variations, bloom formation and growth roles. The absorption or accumulation of fenitrothion by algae in water can contribute to the rapid decrease of fenitrothion levels in water, as well as to the transfer of residues to shellfish and other animals which consume algae.

Fenitrothion is chemically stable in marine waters, even at pH 8.3. The persistence of fenitrothion in these waters is affected mostly by water quality and light, partially by temperature and very little by suspended solids and vapourization. In addition, microbial degradation in marine water is largely suppressed by high salt concentration; a finding supported by determination of higher residual amounts of fenitrothion in marine water near an air-sprayed area compared to adjacent river water levels[19].

There is little information relating to pesticide residues in the marine environment off the Canadian Atlantic coast. On the basis of available information there is no reason to believe that the New Brunswick forest spray program represents any danger to this ecosystem. It should be remembered, however, that the sublethal effects of fenitrothion, aminocarb and their breakdown products have not been well documented.

COMMERCIAL CHEMICALS

The limited size of the manufacturing and industrial sector in the Atlantic Region results in substantially lower volumes of synthetic organic chemicals being released to the environment compared to quantities recorded for more industrialized areas of North America. Nevertheless, there are a few chemical classes that warrant discussion, even though their regional significance remains unclear.

Phthalic Acid Esters

Phthalic acid esters (PAEs) are a group of anthropogenic organic compounds which are frequently detected in trace amounts in environmental samples. They have a variety of uses but are employed principally as plasticizers. In 1973, quantities used in Canada approximated 35.6×10^6 kg[20], but decreased to 27.9×10^6 kg in 1979[2]. Di-(2-ethylhexyl) phthalate (DEHP) is the most widely used PAE with the dibutyl (DBP) and diethyl (DEP) isomers also having numerous applications. The two most important modes of entry of these materials into the Canadian environment are: (1) release from production and processing activities, and (2) release from use and disposal activities[20]. There is no PAE production in the Atlantic Region.

PAEs (DBP, DEP and DEHP) have been detected in the effluents of regional textile plants, pulp and paper mills, oil refineries and electrical manufacturers[22] in the 0.5 µg/L to 100 µg/L range. Most of these industries either discharge directly or in close proximity to an estuary. PAE levels in sediments range between sub-ng/g and low µg/g concentrations[23]. Due to their relative insolubility in water, PAEs are likely to be absorbed to particulate materials and ultimately deposited in sediments. A survey[24] of sediments and waters in 20 harbours in the Maritime Provinces has recently been concluded and results are currently being assessed.

An investigation[25] of dated sediment cores collected in the Western Baltic found initial traces of DEHP in sediments deposited around 1950. A steady increase was observed with maximum concentration of 200 ng/g observed in the youngest layer (0-1 cm) of the core. In the Kiel Bight section of the Baltic[26], phthalate plasticizer concentrations (DEHP, DBP, di-isobutyl phthalate) in the water column ranged between 0.3-203 ng/L.

There is a paucity of information on the bioaccumulation of phthalate esters in the marine environment, with only a few aquatic organisms having been studied[23]. Zitko[27,28] detected DBP in the eggs of double-crested cormorants *(Phalacrocorax auritus)* and herring gulls *(Larus argentatus)* in levels from 11-19 µg/g lipid. He also found DEHP at 13-16 µg/g lipid in the blubber of a common seal pup *(Phoca vitulina)*. DEHP and DPB were measured in commercial fish food at 8 µg/g, respectively. Musial[29] identified di-*n*-hexyl phthalate in Atlantic herring *(Clupea harengus harengus)* and Atlantic mackerel *(Scomber scombrus)* captured off the Nova Scotia coast. Concentrations of this isomer (17-27 µg/g), which is not a major industrial PAE, far exceed (by a factor of approximately 4) those of DBP and DEHP, which are more commonly used in commerce. This indicates a possibility of selective bioaccumulation of di-*n*-hexyl phthalate. Musial[30] found that DEHP in Northwest Atlantic plaice, eel, redfish, herring and mackerel muscle tissues ranged from traces to 10 µg/g. DBP levels in herring and mackerel muscle were 1-2 orders of magnitude lower than DEHP. Morris[31] analyzed specimens of the deep sea jellyfish *(Atolla)* from the North Atlantic and found phthalic acid to be 13% of the total lipid weight and 26% of the total fatty acid weight. These concentrations are considered to be a result of accumulation. Residue concentrations of DBP in fish from several areas of North America have ranged from non-detectable to 0.5 µg/g, and for DEHP levels as high as 3.2 µg/g have been reported[20]. PAE residues in aquatic species from the Great Lakes varied from trace to 1.3 µg/g.

It appears that the rate of PAE clearance is relatively rapid for all biological systems investigated thus far. Mayer[32] exposed fathead minnows to 1.9 µg/L of DEHP for 56 days and found that residues reached an equilibrium concentration of 2.6 µg/g within 28 days. The resulting accumulation factor is nearly 1,400, which agrees with data for DEHP in bluegills[33] exposed to 0.1 µg/L. After removal of the species from contaminated water, 50% of the PAE residues are eliminated within 3-7 days, and after 10 days nondetectable levels are sometimes attained.

The chronic toxicities of PAEs have not been well defined. Laughlin[34] reported that, for the grass shrimp *(Palaemonetes pugio)*, DEHP concentrations up to 1 ppm were without effect on either survival or developmental rate of the larvae. DBP was acutely toxic at 10 to 50 ppm and dimethyl phthalate at concentrations approaching 100 ppm. This investigation and other complementary data suggest that the concentrations reported for the marine environment are well below levels that could possibly warrant environmental concern.

In summary, data on PAEs in marine matrices is limited. Without such information, trends, if they exist, cannot be evaluated. There appears to be PAE accumulation in some Northwest Atlantic fish stocks at levels comparable to those detected in other marine ecosystems.

Organophosphates

Organic esters of phosphoric acid are a class of compounds that are used commercially on a large scale as components in lubricants, fire retardant agents, engine oil, gasoline additives, lubricant bases and plasticizers. Commercial preparations are usually not one particular phosphate ester, but rather mixtures of different isomers. In 1978, it was estimated that 1.4×10^6 kg of industrial organophosphates were sold in Canada[35].

While these materials are structurally similar to the organophosphate pesticides, they are not as readily degradable[36]. Triaryl phosphates (TAPs) are the most used derivatives, and these compounds have been estimated to have a significant residual lifetime in water[37]. The low solubility of TAPs and their high specific gravity (greater than $1 g/cm^3$) make aqueous discharge and spills of such materials environmentally deleterious since sinks of such materials on and in marine sediments would affect the water body for a considerable period of time.

Little regional information on TAPs or other organophosphates, particularly levels in the marine environment, is available. A study of 20 Maritime harbours conducted in the summer of 1980 did not detect TAP residues in either the water column or sediment[38]. Contamination problems further offshore seem unlikely.

Aromatic Amines

Aromatic amines are used primarily as dyes and dye carriers in the textile industry. The carcinogenicity of a number of industrial amines, such as 1- and 2-naphthylamine, has stimulated interest in investigating possible environmental contamination by this class of compounds[39].

Fish samples obtained from U.S. rivers adjacent to nine textile and dyestuff manufacturers were analyzed for aromatic amine content. 1-

Naphthylamine, N-ethyl-N-phenylbenzylamine and N-ethyl-N-(*m*-tolyl) benzylamine were detected in a number of samples[40]. Although the regional textile industry is reasonably active, the extent of aromatic amine discharge and environmental accumulation has not been investigated.

Organotins

Organotins have a variety of commercial applications. One use that is particularly relevant to the Atlantic Region is their use in anti-fouling agents. Bis-(tri-*n*-butyl) tin oxide (TBTO) is commonly used along the Northumberland Strait as a coating for lobster traps to prevent damage by shipworms and wood borers. Toxicity testing[41] has shown TBTO to be extremely toxic to fourth stage lobster larvae (96 h LC50 of 1-10 ppb). The waters and sediments of 10 New Brunswick harbours bordering the Gulf of St. Lawrence were surveyed in the summer of 1981 and samples collected are to be analyzed for TBTO content[38]. Those data will be the first indication as to the status of organotins in the local marine environment.

Organic Arsenicals

Arsenic bonds covalently in both the trivalent and pentavalent states with various organic species to form stable compounds (arsenicals). A number of arsenicals are registered for use in Canada as pesticides and others are utilized in manufacturing and processing activities. The toxicity of arsenicals varies with (1) the chemical nature of the nonmetallic substituent, (2) the species involved, and (3) solubility.

In Atlantic Canada, combustion of fossil fuels may release considerable quantities of arsenic to the air which may ultimately settle in the adjacent marine environment in an organically complexed form. Lisk[42] reported 2-25 ppm of arsenic in coal and 0.05-1.1 ppm in petroleum. However, inorganic forms of arsenic have been found to prevail in natural waters[43] and, consequently, organic arsenicals contribute sparingly to the overall arsenic load. Little information is available on speciation of organic arsenicals in environmental matrices.

SUMMARY

The information base for synthetic organic materials (exclusive of chlorinated compounds and polynuclear aromatic hydrocarbons) in the regional marine environment is limited. With the exception of some phthalic acid esters, reported distributions appear to be considerably less than concentrations of more intensively investigated pollutants such as PCBs and DDTs. No potential environmental problems which might be attributed to such materials have been documented.

REFERENCES

1. Pionke, H.B. and G. Charles. 1973. Pesticide-sediment-water interactions. J. Environ. Qual., 2(1):29-45.

2. New Brunswick Department of Environment. 1981. Pesticide Usage in New Brunswick. Unpublished MS.

3. Ernst, B., G. Pelly and G. Julien. 1981. An examination of two estuaries in Prince Edward Island for contamination by Endrin, MCPA, and Maneb. EPS Surveillance Report, EPS-5-AR-81-7.

4. Christie, P.M.P. and G.H. Penny. 1972. Preliminary results of bioassays to determine the toxicity of some agricultural pesticides used in the Maritimes area. Environ. Can. Res. Dev. Br. M.S. Report 72-2, Halifax, N.S.

5. Zitko, V., D.W. McLeese, W.G. Carson and H.E. Welch. 1976. Toxicity of Alkyldinitrophenols to some aquatic organisms. Bull. Environ. Cont. and Toxicol., 16(5):508-515.

6. Weis, P. and J.S. Weis. 1979. Congenital abnormalities in estuarine fishes produced by environmental contaminants. Anim. Monit. Environ. Pollut. Symp. Pathobiol. Environ. Pollut., Anim. Models Wild. L. Monit., 94.

7. Brown, A.W.A. 1978. Ecology of Pesticides. John Wiley and Sons, Toronto, Ont.

8. Kingsbury, P.D. and B.B. McLeod. 1978. Studies on the environmental impact of aminocarb on aquatic and terrestrial forest ecosystems. p. 105-117. *In:* Proceedings from the Symposium on Aminocarb - Effects of its Use on Environmental Quality. Université de Moncton, Moncton, N.B.

9. Sundaram, K.M.S. and S.Y. Szeto. 1979. A study of the lethal toxicity of aminocarb to fresh water crayfish and its *in-vivo* metabolism. J. Environ. Sci. Health, B14(6):589-602.

10. Robinson, B.H. and F.M. Fisher. 1978. Carbofuran metabolism by the bivalve *Elliptio* and its hemolymph. *In:* American Association for the Advancement of Science: 144th National Meeting, Washington, D.C., Feb. 12-17, 1970. p. 123.

11. McLeese, D.W., D.B. Sergeant, C.D. Metcalfe, V. Zitko and L.E. Burridge. 1980. Uptake and excretion of aminocarb, nonylphenol and pesticide diluent 585 by mussels *(Mytilus edulis)*. Bull. Environ. Contam. Toxicol., 24:575-581.

12. Lord, D.A., R.A.F. Matheson, L. Stuart, J.J. Swiss and P.G. Wells. 1978. Environmental monitoring of the 1976 spruce budworm spray program in New Brunswick, Canada. ESP-5-AR-78-3.

13. McLeese, D.W., V. Zitko, C.D. Metcalfe and D.B. Sergeant. 1980. Lethality of aminocarb and the components of the aminocarb formulation to juvenile atlantic salmon, marine invertebrates and a fresh water clam. Chemosphere, 9:79-82.

14. Ernst, B., G. Julien, K. Doe and R. Parker. 1980. Environmental Investigations of the 1980 Spruce budworm spray program in New Brunswick. EPS Surveillance Report, EPS-5-AR-81-3.

15. Symons, P.E.K. 1973. Behaviour of young atlantic salmon *(Salmo salar)* exposed to or force-fed fenitrothion, an organophosphate insecticide. J. Fish. Res. Board of Can., 30:651-655.

16. Mallet, V.N. 1977. Analytical results for fenitrothion in shellfish. Report No. 8. Université de Moncton, Moncton, N.B.

17. McLeese, D.W., V. Zitko and D.B. Sergeant. 1979. Uptake and excretion of fenitrothion by clams and mussels. Bull. Environ. Contam. Toxicol., 22:800-806.

18. Lakshminarayana, J.S.S. and H. Bourque. 1980. Absorption of fenitrothion by plankton and benthic algae. Bull. Environ. Contam. Toxicol. 24:389-396.

19. Kodama, T. and S. Kuwatsuka. 1980. Factors for the persistence of Parathion and Fenitrothion in sea water. J. Pest. Sci. 5(3):351-355.

20. Leah, T.D. 1977. The production, use and distribution of phthalic acid esters in Canada. Environmental Contaminants Inventory Study, No. 4. IWD Report Series No. 47. Ont.

21. Leah, T.D. Environmental Protection Service, Ottawa, Ont. Personal Communication. 1979.

22. Plansearch. 1981. Chlorinated phenols/phthalate esters in industrial effluents. Report for Environment Canada.

23. Pierce, R.C., S.P. Mathur, D.T. Williams and M.J. Boddington. 1980. Phthalate esters in the aquatic environment. National Reasearch Council of Canada. NRCC Report No. 17583.

24. Belliveau, P. Environmental Conservation Service, Inland Waters Directorate, Moncton, N.B. Personal Communication. 1981.

25. Muller, G., J. Dominik and R. Reuther. 1980. Sedimentary record of environmental pollution in the Western Baltic Sea. Naturwissenschaffen, 67:595-600.

26. Ehrhardt, M. and J. Derenbach. 1980. Phthalate esters in the Kiel Bight. Marine Chem. 8:339-346.

27. Zitko, V. 1972. Determination, toxicity and environmental levels of phthalate plasticizers. J. Fish. Res. Board of Can., Tech. Rep. No. 344, St. Andrews.

28. Zitko, V. 1973. Determination of phthalates in biological samples. Intern. J. Anal. Chem., 2:241-252.

29. Musial, C.J., J.F. Uthe, G.R. Sirota, B.G. Burns, M.W. Gilgan, V. Zitko and R.A. Matheson. 1981. Di-n-hexyl Phthalate (DHP), a newly identified contaminant in Atlantic herring *(Clupea harengus harergus)* and Atlantic mackerel *(Scomber scrombrus)* Can. J. Fish. Acquatic Sci., 38: 856-859.

30. Musial, C.J. and J.F. Uthe. 1980. Studies on phthalate esters in Western Atlantic Finfish. ICES Paper, C.M. 1980/E:11.

31. Morris, R.J. 1970. Phthalic acid in the deep sea jellyfish *(Atolla)*. Nature, 227:1264.

32. Mayer, F.L. and H.O. Sanders. 1973. Toxicology of phthalic acid esters in aquatic organisms. Environ. Health Perspect, 3:153-158.

33. Johnson, B.T. 1975. Unpublished data. Fish-Pesticide Research Laboratory, Columbia, Missouri.

34. Laughlin, R.B. Jr., J.M. Neff, Y.C. Hrung, T.C. Goodwin and C.S. Giam. 1978. The effects of three phthalate esters on the larval development of the grass shrimp *(Palaemonetes pugio)* (Nolthuis). Water, Air and Soil Poll.: 323-336.

35. Martec Limited. 1979. Canadian use pattern survey of organotins, phthalate esters and triaryl phosphates. Volume C. Internal Report for Environmental Protection Service.

36. Uthe, J.F. 1976. Triaryl phosphates as environmental concerns. ICES Paper, C.M. 1976/E:3.

37. Wagemann, R.B. Graham and L.W. Lockhart. 1974. Studies on chemical degradation and fish toxicity of a synthetic triaryl phosphate lubrication oil, IMOL S-140. Fish. Mar. Serv. Tech. Rep., No. 486.

38. Belliveau, P. Environmental Conservation Service, Inland Waters Directorate, Moncton, N.B. Personal communication. 1982.

39. Stender, J.N. 1974. Occupational Safety and Health Admin., Dept. of Labour. Part 1910 - Occupational Safety and Health Standards - Carcinogens. Fed. Regist., 39(20):3756.

40. Diachenko, G.W. 1979. Determination of several industrial aromatic amines in Fish. Env. Sci. and Tech. 13(3):329-333.

41. Parker, W.R. 1978. The acute lethality of Bis-tri-n-butyl tin oxide to lobster larvae *(Homarus americanus)*. Unpublished MS.

42. Lisk, D.J. 1972. Trace metals in soils, plants and animals. Adv. Agron. 24:267-325.

43. National Research Council of Canada. 1978. Effects of arsenic in the Canadian environment. NRCC Report No. 15391.

A PERSPECTIVE
ON POTENTIAL ENVIRONMENTAL PROBLEMS ASSOCIATED WITH CHLORINE USAGE

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"For in much wisdom is much vexation, and the more a man knows, the more he has to suffer - Ecclesiastes 1:18."

INTRODUCTION

The role of contaminated water in the spread of infectious disease is believed to have been well understood in biblical times, yet it is only recently that disinfectants such as chlorine have been employed on a wide scale to combat potential waterborne diseases. As far back as 2000 BC, the value of exposing foul water to heat and sunlight was recognized, while there are records from later ages indicating that quicklime and copper salts were used for disinfection and sterilization purposes. By the turn of the 19th Century, chlorine was being employed as a sewage disinfectant in some districts of London, and the chemical began to be used shortly thereafter in the United States and western Europe for treating both sewage effluents and potable water. The benefits of using chlorine for water treatment are manifest since the incidence of waterborne intestinal diseases, particularly typhoid fever and cholera, has declined precipitously in the last 80 years.

The use of chlorine in potable water treatment has expanded to include, in addition to disinfection, several other important functions such as taste, color, and odor removal, iron and manganese removal, prevention of water quality degradation in distribution systems, and sterilization of water mains and reservoirs[1]. Chlorine has also found use in the food industry for various purposes. For instance, it is often used at high concentrations in slaughter houses to disinfect meat and poultry products, and in the milling industry certain types of flour are chlorinated in order to enhance maturation rate and overall chemical-physical characteristics. In this regard, it is of interest to note that approximately 1.5% of the total flour produced in the United Kingdom is chlorinated, the average level of chlorination being very high, about 1250 ppm[2].

CHLORINE USAGE

The annual production of chlorine in North America is estimated at 10.5 million tonnes. Only about 20% of this amount is of direct interest since about 80% of the total is used by industry as chemical feedstock for the production of such products as plastics, synthetic fibers and solvents. Considering the 20% of interest, the pulp and paper industry is the major user, accounting for about 16%, while the remaining 3-4% is principally

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used for water treatment. Much smaller quantities are used in the food industry. The amounts of chlorine used in Canada for various purposes[3] are summarized in Table 1.

CHLORINE CHEMISTRY

When chlorine is added to seawater, it rapidly hydrolyzes to form hypochlorous acid and chloride according to the reaction,

$$Cl_2 + H_2O \rightarrow HOCl + HCl$$

Hypochlorous acid (HOCl) is regarded as the major reactive species for most oxidizing reactions of aqueous chlorine between pH 5-9. The acid acts as an electrophilic reagent with the chlorine atom taking on the characteristics of a Cl^+ species which can react with a variety of natural compounds and form a wide array of chloroorganics[4]. When chlorine or hypochlorite is added to seawater, which (naturally) contains relatively high concentrations of bromide ions, there is a rapid formation of hypobromous acid[5-7] according to the reaction:

$$Br^- + HOCl = HOBr + Cl^-$$

The resulting hypobromous acid is also an electrophilic species, and is actually more reactive than its sister compound, hypochlorite. Water containing high concentrations of ammonia may result in rapid formation of chloramines (fresh water) or bromamines (salt water)[8], both of which are oxidative species and display a significant level of acute toxicity.

TABLE 1 - CHLORINE PRODUCTION AND CONSUMPTION IN CANADA

	1973	1975	1976
Total production (metric tonnes)	889,948	748,894	898,336
Total consumption of industries	818,330	604,787	792,815
Consuming Sectors	**% Consumption by Industries Surveyed**		
Municipal waterworks	0.7	1.0	-
Mining	0.6	0.8	-
Pulp and paper	57.3	83.5	62.4
Smelting and refining	0.9	1.1	-
Soap and cleaning compounds	1.1	1.2	0.9
Industrial chemicals	39.4	12.0	36.6
Fish processing	-	-	0.02
Miscellaneous food industries	-	-	0.01
Miscellaneous	0.1	0.3	0.07
	100.0	100.0	100.0

RECENT CONCERNS ABOUT CHLORINE USAGE

Benefits associated with the use of chlorine as a disinfectant and antifouling agent are widely recognized; nevertheless, during the last few years evidence has begun to appear suggesting that the wholesale use of chlorine could have a health and environmental cost. The acute toxicological effects on aquatic organisms of such reactive species as chlorine, hypochlorous acid and chloramines have been recognized for some time[9-11] but concern has recently begun to focus on the formation of carcinogenic / mutagenic (and putatively teratogenic) compounds in chlorinated water[12-15]. It has also been shown that a large variety of complex chloroorganic compounds may be produced when chlorine is applied to organic rich waters[16-18]. As indicated in Fig. 1, some of these compounds are identical to or not unlike many compounds listed as priority contaminants by regulatory agencies in the U.S.A. and Canada. The formation of such a variety of compounds has especially led to questions concerning the long term impact of chlorination derived chemicals on human and environmental health.

DRINKING WATER

The mutagenesis aspect of chlorination has only recently begun to unfold but there has been a special interest in water chlorination since it was revealed that chloroform, a regulated carcinogen, could generally be found in chlorine treated drinking water. Survey studies in the U.S.A.[19], Canada [20], and Europe[21] have demonstrated that chloroform and lesser quantities of

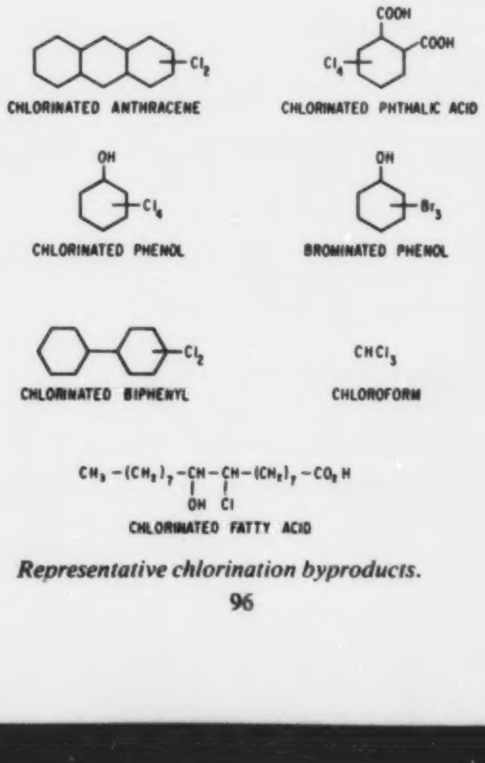

COOH
COOH
Cl2
Cl4
CHLORINATED ANTHRACENE
CHLORINATED PHTHALIC ACID
OH
Cl4
OH
Br3
CHLORINATED PHENOL
BROMINATED PHENOL
Cl2
CHLORINATED BIPHENYL
CHCl3
CHLOROFORM
CH3 - (CH2)7 - CH - CH - (CH2)7 - CO2H
OH Cl
CHLORINATED FATTY ACID

Figure 1: Representative chlorination byproducts.

other volatile trihalomethanes, including the recently recognized mutagens, dibromochloromethane, dichlorobromomethane, and bromoform are commonly associated with water chlorination (Table 2).

TABLE 2 - TRIHALOMETHANE CONTENT OF DRINKING WATER

Compound	Median Concentration (µg/L)		
	USA	Canada	Germany
Chloroform	21.0	22.7	6.4
Dichlorobromomethane	6.0	2.9	3.4
Dibromochloromethane	1.2	0.4	-
Bromoform	5.0	0.1	-

Halogenated hydrocarbons are known to be ubiquitous in water supplies where up to 30% of the identified compounds have been shown to be chlorinated or brominated derivatives. Within the last few years it has become apparent that water chlorination is not only responsible for the production of volatile haloforms, many of which are known mutagens and carcinogens, but also for the production of "non-volatile", high molecular weight mutagenic substances. In reference to human health, recent epidemiology studies in the U.S.A. have indicated that there may be a slight risk of carcinogenesis associated with the consumption of chlorinated drinking water[22-27]. It has been noted, however, that the evidence for such an association is still tenuous.

The toxicological potential of drinking water is related to the amount of chlorine used for disinfection and this will depend on the quality of the raw water supply as well as other factors such as concentration of nutrients, temperature, and pH. The concentration of chlorine commonly used to disinfect drinking water is in the 1-16 mg/L range, the higher concentrations being employed to treat low quality water[1]. For instance, the Seine River in Paris is treated with 16 mg/L of chlorine and a comparable low quality water in Canada is the Grand River at Brantford, Ontario. By contrast, the drinking water supplies of such large cities as Chicago and San Francisco require less than 1 mg/L of chlorine. It is noteworthy that the higher chlorine dosages are comparable to the levels used for sewage treatment[28]. It is unlikely that "sewage level" concentrations of chlorine are required to disinfect water supplies in the Atlantic Provinces, but town-by-town documentation is required before "definitive" statements can be made. It is of special interest, however, that humic and fulvic acids have been shown to be important precursors for the formation of haloforms such as chloroform[29,30]. Elevated concentrations of both humic and fulvic acids are common in soft water and this may be of some concern for many water supplies in the Atlantic Provinces.

AQUATIC ENVIRONMENTAL TOXICITY

To date, aquatic environmental concerns about chlorine usage have principally focussed on the presence of toxic chlorine residuals in effluents from power plants, pulp mills, and sewage treatment facilities. Residual chlorine or those compounds which possess an oxidative capacity, such as chloramines, are primarily responsible for acute effects, while chronic effects are presumably mediated by trace amounts of residual chlorine as well as the variety of novel chloroorganics (and other compounds) produced by chlorination. A considerable amount of data has been collected to indicate that chlorine and its reaction products are toxic, in very low concentrations, to many types of fresh water, estuarine and marine organisms. Summary data on chlorine toxicity to both fresh water and marine organisms are portrayed in the modelling studies of Mattice and Zittel[31] (Fig. 2). It is of interest that effluent guidelines for chlorine discharge to fresh water environments in the United States were implemented by the Environmental Protection Agency on the basis of the Mattice-Zittel fresh water model.

Toxicity data have prompted the setting of total residual chlorine criteria to protect fish and aquatic life. The levels proposed by the U.S. National Academy of Sciences[32], the European Inland Fisheries Advisory Commission[33] and the U.S. Environmental Protection Agency[34] range from 0.002 mg/L to 0.004 mg/L for continuous exposure of the most sensitive organisms. Similar recommendations are stated by the Great Lakes Water Quality Board of the International Joint Commission[35]. The figure of 0.002 mg/L seems to best represent the criteria developed by the majority of regulatory agencies for maximum safety.

PAPER MILLS

Use of chlorine by the paper industry has received considerable attention since large amounts of the chemical are employed in some processes of pulp bleaching in order to satisfy our demand for clean, white paper. The total consumed by the paper industry actually amounts to about ten times that used for water and sewage treatment and, whereas water and sewage treatment involves usage of small amounts of chlorine at hundreds of municipalities, pulp mills employ large dosages of the chemical at relatively few point sources. Effluents from such point sources often drain into rivers or coastal waters creating concern for potential effects on aquatic life. However, pulp mill effluents are constantly being improved through in-plant changes and use of treatment ponds and such modifications should be effective in reducing the levels of most compounds of concern.

The high concentrations of chlorine that are required and the acid conditions of pulp bleaching would be expected to increase both the number and stability of novel chloroorganics in untreated pulp mill effluents. It has been estimated that about 10% of the chlorine applied to pulp is incorporated into non-volatile organic compounds while considerably more may be incorporated into volatile organics such as chloroform. Loading of

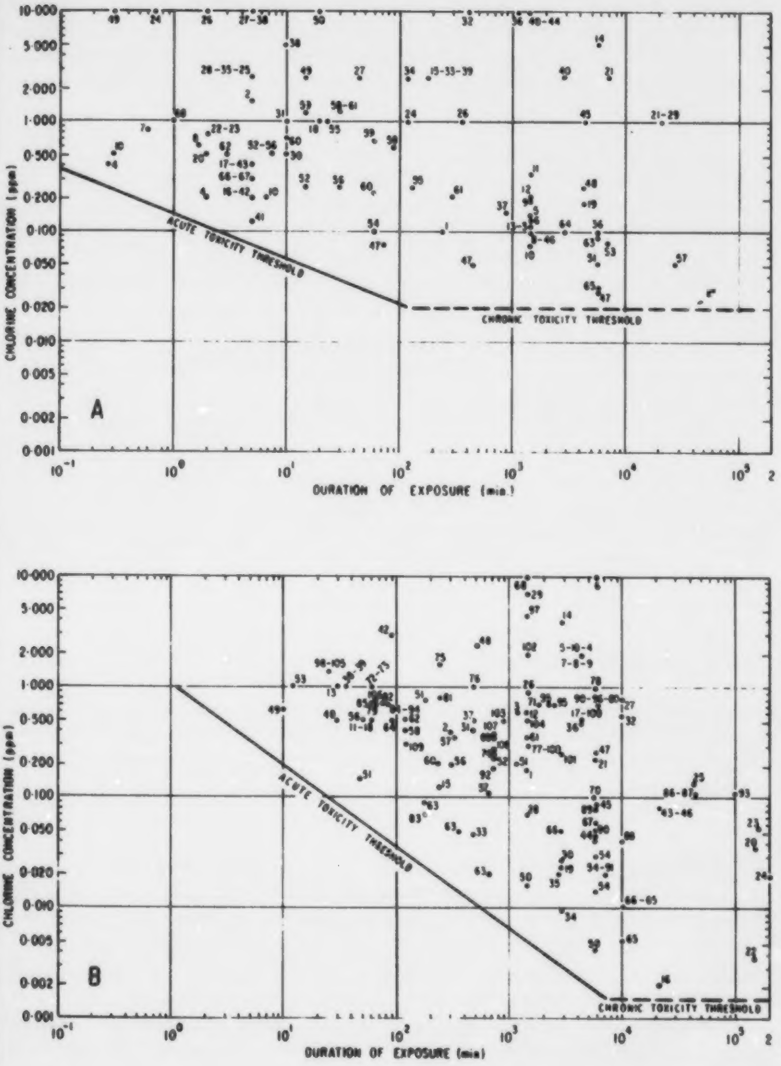


Figure 2: *(a) and (b) Toxicity of chlorine to marine organisms: points represent median effect. levels for mortality or sublethal effects.*

organically bound chlorine from kraft-mill bleach plants is typically 3-8 kg/tonne of pulp[36,37] and the chlorine content of dissolved organic material extracted from chlorination stage effluents is 20-30%. This indicates that untreated effluents from a 1000 tonnes/day pulp mill could contain 10-40 tonnes/day of chlorinated organics[38].

Scandinavian workers[39,40] were the first to establish the importance of chlorination in the production of mutagenic compounds in pulp mill effluents. Subsequent studies in a number of laboratories have confirmed these original observations and it is presently believed that the use of chlorine in pulp bleaching has the potential to produce a variety of mutagenic compounds. Many of these mutagens may be halogenated derivatives but other candidate compounds could be aldehydes and ketones produced as oxidation by-products of the bleaching process.

The pulp mills located in Newfoundland do not utilize chlorine for pulp bleaching but it is used in some mills in Nova Scotia and New Brunswick (Fig. 3). The kraft mills in Newcastle, Saint John, Nackawic, and New Glasgow each produce approximately 580 tonnes of pulp per day, using a total of about 200 tonnes of chlorine. The three sulfite mills situated at Atholville, Edmundston, and Point Tupper produce a total of 1000 tonnes of pulp per day, using about 55 tonnes of chlorine. All in all, it would appear that untreated effluents from these mills have the potential to inject several thousand tonnes of chlorinated organics into the aquatic environment yearly. Chlorine mediated acute toxic effects might be expected near pulp mill outfall sites but effective dilution by receiving waters should generally limit any impacted zones. The possible sublethal effects of trace amounts of chlorinated hydrocarbons and other by-products of the chlorine bleaching process would seem to be more important, both from an environmental and public health perspective. In this regard any fish, and especially shellfish species, found in the general area of pulp mill discharges should be investigated for the presence of chlorination derived organohalogens.

SEWAGE

Sewage chlorination is a widely accepted practice of wastewater treatment in Canada, including the Maritimes. Most sewage effluents are directed into fresh water and marine environments and this can potentially provide a large number of point sources of chloroorganics. Similar to studies on chlorine bleached pulp, a large variety of halogenated derivatives, including substituted purines, pyrimidines, phenols, and aromatic acids have been identified in sewage sludge. Addition of chlorine to organic rich sewage waters is also known to be quite effective in generating mutagenic compounds.

Studies on wastewater treatment have indicated that reaction yields of chloroorganic products in chlorinated sewage effluents range from 0.5 to 3.0%, or higher. The Environmental Protection Agency has recently gathered total organic chlorine (TOCl) data for 50 raw and finished waters in the United States[41]. The range of TOCl values for raw water was less (5-122 µg/L) than for finished water (14-310 µg/L). Information on the levels of TOCl found in representative sewage waters is not available, but it is of interest that the range of chlorine levels used for sewage disinfection is not unlike the levels used to treat low quality drinking water. However, even if it is assumed that TOCl concentrations in the 200-300 µg/L range, the upper limit noted in the EPA studies, are commonly found in sewage water, the total input of chloroorganics from sewage would be quite small compared with potential inputs from pulp mills. It has been calculated from data contained in the National Inventory of Municipal Waterworks and Wastewater Systems[42], that approximate daily discharges of chlorinated sewage from Prince Edward Island, Newfoundland, Nova Scotia, and New Brunswick are 17, 25, 35 and 44 million litres, respectively. Assuming an estimate of 200 µg of "soluble" chlorinated derivatives formed per litre, this would translate into approximate yearly discharges of 1.4, 2.0, 2.8 and 3.6 tonnes of chloroorganics.

In reference to direct ecological effects, caution is especially warranted where effluents from large treatment plants enter small streams and rivers which do not provide effective dilution. Residual chlorine in the effluent from municipal treatment plants has been implicated in fish kills at various places in the U.S.A.[43,44], and studies in British Columbia[45] have reported acute toxicity or absence of salmonids in streams receiving 0.02 mg/L more

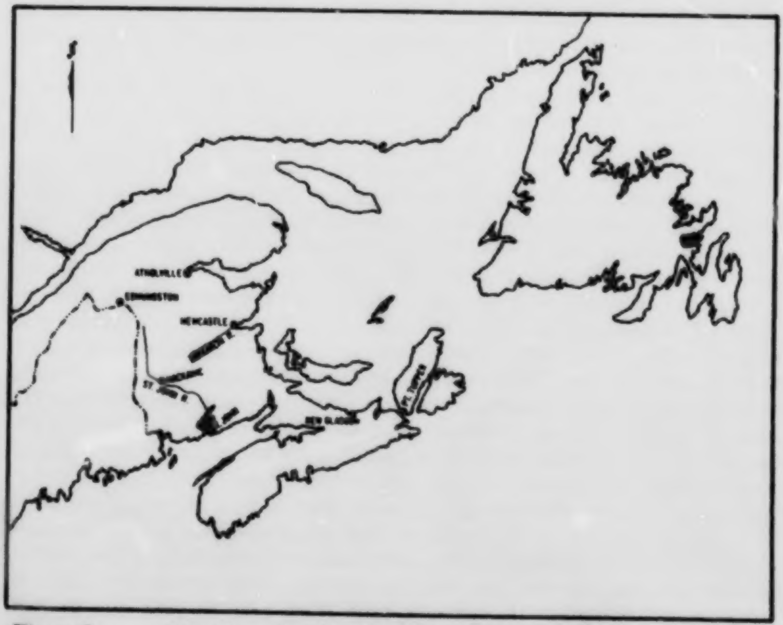


Figure 3: Location of pulp mills that use chlorine bleaching.

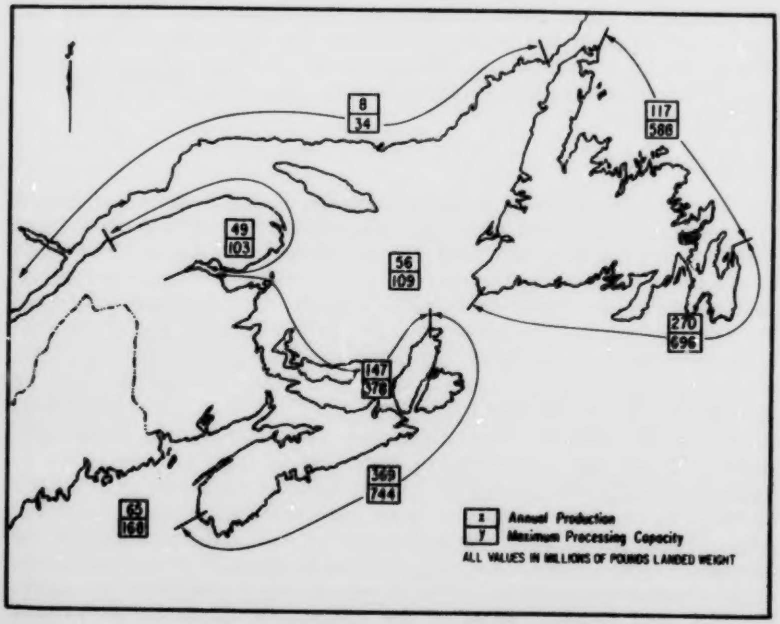
of chlorine. One hundred and forty-nine domestic wastewater plants were studied in Virginia, Pennsylvania, and Maryland and it was observed that both community diversity and fish occurrence were related to chlorine concentrations[46]. In chlorine plume surveys conducted downstream from nine treatment plants in New Jersey, no fish were observed at chlorine concentrations greater than 0.11 mg/L and, in general, fish were not observed until chlorine concentrations reached greater than 0.05 mg/L. However, it should also be appreciated that localized impacts could be "offset" by sewage mediated increases in productivity at points downstream. It was also noticed in the New Jersey study that schooling fish would actually avoid the chlorinated plume. This may be of special importance if chlorinated sewage effluents drain into the estuarine portions of rivers which provide habitat for migrating species such as salmon. Notwithstanding the direct effects of chlorine on such important migrating species as salmon, any acute effects associated with wastewater treatment should be more or less localized, and the potential for contamination of commercial fish species, such as shellfish, with wastewater derived chloroorganics is probably of greater concern.

FISH PLANTS

Chlorine is generally chosen for usage in the food industry in Canada for disinfection and general sanitary purposes. Freshly processed meats and poultry are also treated with chlorine in some countries, but this practice is not permitted in Canada. Of particular interest, however, is the practice in Canada of using chlorine in fish plants to sanitize processing waters. About 8 litres of water are utilized per pound of fillet and the average quantity of fillets produced yearly by fish plants in the Maritimes and Newfoundland is approximately 400 million pounds, with production in both areas being approximately comparable. Up to 5 ppm residual chlorine may be used on filleting lines while 10 ppm may be used for washing round fish. At these concentrations, the chlorination yield of organohalogens might be expected to be high. However, even if it is assumed that fish processing and plant sanitation waters are similar to low quality drinking water and could generate up to 200-300 µg/L of TOCl, the total level of chioroorganics produced would still be quite low, compared with untreated pulp mill effluent. For instance, assuming an estimate of 200 µg of TOCl formed per litre of water, this would translate into approximate yearly discharges of 0.35 tonnes each from Newfoundland and the Maritimes. In terms of input, the volume of chloroorganics originating from fish plants may not be high, but the quantity seems significant, and should be included in the overall budget of chlorination derived organhalogens which enter the aquatic environment. It is also of interest to note that regulations require a lower residual of chlorine for sewage treatment than is used in processing fish. The higher chlorine dosages used in fish plant operations may produce compounds which are more highly substituted with chlorine and thus have a greater potential for environmental persistence, than sewage chlorination by-products. Although fluctuations exist from year to year due to fish landings and plant closures, the primary areas of fish plant activity in Newfoundland and the Maritimes are indicated in Fig. 4[47].

ENVIRONMENTAL PERSISTENCE

Environmental degradability is one of the most important factors to be considered when evaluating the health and ecological effects of chemicals. Compared to such refractory chlorinated hydrocarbons as PCBs and DDTs, it is unlikely that most of the organochlorines formed by water chlorination persist in the environment to any significant degree. In agreement with chemical theory, it has been shown that many of the substances of concern, such as chlorinated biphenyls and aromatics are not highly substituted, and compounds having a low level of chlorine substitution, are generally considered to be environmentally labile. In support of the concept of rapid degradability, one field study has reported that water chlorination derived compounds isolated from Biscayne Bay, Florida, were quite labile. Chlorinated fatty acids found in chlorinated flours are also known to be effectively metabolized by mammalian enzyme systems. However, there is still a virtual lack of data on the relative persistence of various types of chlorination derived organics, and it is not known if the degradation rates of these chemicals is measurable in days, weeks or


8
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ALL VALUES IN MILLIONS OF POUNDS LANDED WEIGHT

Figure 4: *Distribution and capacities of freezing plants.*

months. Bioaccumulation and persistence in various species of animals similarly warrants some attention.

MAN AND NATURE AS SOURCES OF TOXIC ORGANOHALOGENS

Water chlorination has the potential to produce a wide variety of chloroorganics, but on a worldwide scale it has to be appreciated that the aquatic environment itself could be a major source of toxic organohalogens, many of which are known mutagens or carcinogens (reviewed by Payne and Rahimtula[44]). Toxic compounds known to be produced by marine plants inlcude a number of haloforms common to water chlorination, including the mutagens bromoform, dibromochloromethane and dichlorobromomethane as well as the well recognized carcinogen, chloroform. Also, a variety of mutagens may be found amongst many of the high molecular weight organohalogens common to marine plants; five polyhalogenated compounds were recently isolated from a marine alga source and all were found to be mutagenic. Numerous other halogenated compounds, many with structures analogous to those of known carcinogens and other toxicants are found in both marine and terrestrial organisms. Recent observations that many compounds in the natural aquatic environment may have a great potential for effecting such toxicological endpoints as mutagenicity and carcinogenicity are of obvious importance and should be included in attempts to evaluate the benefits and risk of chlorine usage. However, the natural toxicant conundrum does not mean that we should dismiss all concerns for possible health and ecological effects connected with chlorine usage, especially in those processes which have the potential to produce point sources of relatively high concentrations of organohalogens.

CONCLUSIONS

The many benefits of using chlorine for disinfection, defouling and bleaching purposes are widely recognized; nevertheless, during the last few years concern has been expressed that wholesale use of the chemical could have a health and environmental cost. Both chlorine and chlorine residual compounds which contain an oxidative capacity are quite toxic to aquatic animals, and present regulations attempt to ensure sufficient dilution of effluents in order to minimize both acute and subacute effects in receiving waters. A multitude of organohalogen and other compounds may be formed during various chlorination processes and it is these novel compounds which presently require attention. Observations that many compounds formed as by-products of water chlorination are mutagenic in bacterial systems are of special interest since mutagenicity is presently accepted by regulatory agencies as an important toxicological endpoint. Also of importance from a regulatory point of view is the potential for the formation of many organohalogen compounds, many of which may be not unlike some of the compounds listed as priority contaminants by agencies in the U.S. and Canada. Most of the compounds produced by various chlorination processes are suspected to be metabolically and environmentally labile and identical or similar compounds appear to be common in nature. There is a need, however, to evaluate the environmental persistence of chlorination derived compounds and to determine if they accumulate in biota, especially in organisms such as shellfish found near point sources of potential high level contamination. In the Maritimes, major point sources could include pulp mills while minor point sources could include fish plants and sewage treatment facilities.

In summary, there is need for a preliminary assessment on the fate of various chloroorganics discharged from pulp mills, sewage works and fish plants, into aquatic receiving systems in the near-shore environment. Effluent dilution can remedy acute toxicity problems associated with point source discharges of chlorine and chlorine residuals but the production of even small concentrations of relatively persistent compounds would present new concerns for the offshore as well as the near-shore environment.

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REFERENCES

1. White, G.C. 1978. Current chlorination and dechlorination practices in the treatment of potable water, wastewater and cooling water, p. 1-18. *In:* R.L. Jolley (ed.), Water Chlorination: Environmental Impact and Health Effects, Vol. 1, Ann Arbor Science, Ann Arbor, Michigan, 439 p.

2. Neal, R.A. 1980. Known and projected toxicology of chlorination by-products, p. 1007-1017. *In:* R.L. Jolley, W.A. Brungs, R.B. Cumming (eds.), Water Chlorination: Environmental Impact and Health Effects, Vol. 3, Ann Arbor Science, Ann Arbor, Michigan, 1171 p.

3. Health and Welfare Canada. 1982. Municipal wastewater disinfection in Canada: need and application, p. 5. *In:* A Report of the Working Group on Wastewater Disinfection to the Federal-Provincial Advisory Committee on Environmental and Occupational Health, Information Directorate, Dept. National Health and Welfare, Ottawa, Canada.

4. Morris, J.C. 1978. The chemistry of aqueous chlorine in relation to water chlorination, p. 21-35. *In:* R.L. Jolley (ed), Water Chlorination: Environmental Impact and Health Effects, Vol. 1, Ann Arbor Science, Ann Arbor, Michigan, 439 p.

5. Carpenter, J.H. and D.L. Macalady. 1978. Chemistry of halogens in sea water, p. 161-179. *In:* R.L. Jolley (ed.), Water Chlorination: Environmental Impact and Health Effects, Vol. 1, Ann Arbor Science, Ann Arbor, Michigan, 439 p.

6. Middaugh, D.P. and W.P. Davis. 1976. Impact of chlorination processes on marine ecosystems. U.S. Environmental Protection Agency, Office of Research and Development, EPA-60013-76-079, Water Quality Research of the U.S. EPA USA NTIS PB-257-091.

7. Pierce, R.C. 1978. The aqueous chlorination of organic compounds: chemical reactivity and effects on environmental quality, p. 25-34. National Research Council of Canada, Publication No. 16450, 136 p.

8. Inman, G.W. and J.D. Johnson. 1978. The effect of ammonia concentration on the chemistry of chlorinated sea water, p. 235-252. *In:* R.L. Jolley, H. Gorchev and D.H. Hamilton (eds.), Water Chlorination: Environmental Impact and Health Effects, Vol. 2, Ann Arbor Science, Ann Arbor, Michigan, 909 p.

9. Brungs, W.A. 1973. Effects of residual chlorine on aquatic life - a literature review. J. Water Pollut. Control Fed., 45: 2180-2193.

10. Hall, L.W., D.T. Burton and L.H. Liden. 1981. An interpretative literature analysis evaluating the effects of power plant chlorination on fresh water organisms. CRC Critical Reviews in Toxicology, 9(1): 1-20.

11. Davis, W.P. and D.P. Middaugh. 1978. A revised review of the impact of chlorination processes upon marine ecosystems: Update 1977, p. 283-310. *In:* R.L. Jolley (ed.), Water Chlorination: Environmental Impact and Health Effects, Vol. 1, Ann Arbor Science, Ann Arbor, Michigan, 439 p.

12. Loper, J.C., D.R. Lang, R.S. Schoeny, B.B. Richmond, P.M. Gallagher and C.C. Smith. 1978. Residue organic mixture from drinking water show *in vitro* mutagenic and transforming activity. J. Toxicol. Environ. Health, 4: 919-938.

13. Rappaport, S.M., M.G. Richard, M.C. Hollstein and R.E. Talcott. 1979. Mutagenic activity in organic wastewater concentrate. J. Environ. Sci. Technol., 13(8): 957-961.

14. Nestmann, E.R., G.L. LeBel, D.T. Williams and D.J. Kovibel. 1979. Mutagenicity of organic extracts from Canadian drinking water in the *Salmonella* mammalian microsome assay. Environ. Mutag., 1: 337-345.

15. Payne, J.F., I. Martins, D. Fagan and A. Rahimtula. 1980. Mutagens in aquatic environments: water chlorination and petroleum sources of polycyclic aromatics as major contributors, p. 845-850. *In:* R.L. Jolley, W. Brungs, R.B. Cumming (eds.), Water Chlorination: Environmental Impact and Health Effects, Vol. 3, Ann Arbor Science, Ann Arbor, Michigan, 1171 p.

16. Gaffney, P.E. 1977. Chlorobiphenyls and PCBs: formation during chlorination. J. Water Pollut. Cont. Fed., 49: 401-404.

17. Oyler, A.R., D.L. Bodenner, K.J. Welch, R.J. Liukkonen, R.M. Carlson, H.L. Kopperman and R. Caple. 1978. Determination of aqueous chlorination reaction products of polycyclic aromatic hydrocarbons by reversed phase high performance liquid chromatography gas chromatography. Analyt. Chem., 50: 837-842.

18. Snider, E.H. and F.C. Alley. 1979. Kinetics of the chlorination of biphenyl under conditions of water treatment processes. Environ. Sci. Technol., 13: 1244-1248.

19. U.S. Environmental Protection Agency. 1975. Preliminary assessment of suspected carcinogens in drinking water. Report of Congress, Washington, D.C., U.S. Government Printing Office.

20. Health and Welfare Canada. 1977. National survey of halomethanes in drinking water. Environmental Health Directorate, Dept. National Health and Welfare, Ottawa, Canada.

21. Sonneborn, M. and B. Bohn. 1978. Formation and occurrence of haloforms in drinking water in the Federal Republic of Germany, p. 537-542. *In:* R.L. Jolley, H. Gorchev and D.H. Hamilton, Jr. (eds.), Water Chlorination: Environmental Impact and Health Effects, Vol. 2, Ann Arbour Science, Michigan, 909 p.

22. Cantor, K.P., R. Hoover, T. Mason and L.J. McCabe. 1977. Association of cancer mortality rates and trihalomethane levels in municipal drinking water supplies. Am. J. Epidemiol., 106: 230-231.

23. DeRouen, J.A. and J.F. Diem. 1975. The New Orleans drinking water controversy. A statistical perspective, Am. J. Public Health, 65: 1060-1062.

24. Kuzma, R.J., C.M. Kuzma and C.R. Buncher. 1977. Ohio drinking water source and cancer rates. Am. J. Public Health, 67: 725-729.

25. Page, T., R.H. Harris and S.S. Epstein. 1976. Drinking water and cancer mortality in Lousiana. Science (Washington), 193: 55.

26. National Academy of Sciences. 1980. Drinking water and health. Washington, D.C.

27. Council on Environmental Quality. 1980. Drinking water and cancer: a review of recent findings and assessment of risks. Washington, D.C.

28. Ross, S.A., P.H.M. Gou and B.E. Jank. 1980. Design and selection of small wastewater treatment systems. Environmental Protection Service, Environment Canada, Report No. EPS-3-WP-80-3, 396 p.

29. Larson, R.A. and R.L. Rockwell. 1979. Chloroform and chlorophenol production by decarboxylation of natural acids during aqueous chlorination. Environ. Technol., 13(3): 325-329.

30. Oliver, B.G. and S.A. Visser. 1980. Chloroform production from the chlorination of aquatic human material: the effect of molecular weight, environment and season. Water Research, 14: 1137-1141.

31. Mattice, J.S. and H.E. Zittel. 1976. Site-specific evaluation of power plant chlorination. J. Water Pollut. Control Fed., 48(10): 2284-2308.

32. National Academy of Sciences, National Academy of Engineering. 1972. Water Quality Criteria, US-EPA R3-73-033.

33. European Inland Fisheries Advisory Commission. 1973. Water quality criteria for European fresh water fish, report on chlorine and fresh water. Technical Report No. 29 (Rome: FAO, 1973).

34. United States Environmental Protection Agency. 1976. Quality Criteria for Water. US-EPA 440/9-76-023.

35. Great Lakes Water Quality Boards. 1976. Great Lakes water quality. Annual Report, International Joint Commission.

36. Hardell, H.L. and F. deSousa. 1977. Characterization of spent bleaching liquors, Part I - Spent liquors from the chlorine and alkali extraction stages in the prebleaching of pine kraft pulp. Svensk. Papperstidn., 80: 110-120.

37. Lund, J.H., C.R. Cook, H.P. Meier, L.T.K. Chung and J.M. Leach. 1979. Effects of oxygen delignification on kraft mill effluent quality. Canada Department of the Environment, CPAR Report No. 914-1.

38. Leach, J.M. 1980. Loadings and effects of chlorinated organics from bleached pulp mills, p. 325-334. *In:* R.L. Jolley, W.A. Brungs, R.B. Cumming (eds.), Water Chlorination: Environmental Impact and Health Effects, Vol. 3, Ann Arbor Science, Ann Arbor, Michigan, 1171 p.

39. Ander, P., K. Erikisson, M.C. Kolar, K. Kringstod, V. Rannug and C. Ramel. 1977. Studies on the bleaching properties of bleaching effluents. Sven. Papperstidn., 80: 454-459.

40. Bjorseth, A., G.E. Carlberg and M. Moller. 1979. Determination of halogenated organic compounds and mutagenicity testing of spent bleach liquors. Sci. Total Environ., 11: 197-211.

41. Christman, R.F. 1980. Editorial: a perspective on chlorination. Environ. Sci. Technol., 14(1): 5.

42. Supply and Services Canada. 1978. National inventory of municipal wastewater and wastewater systems in Canada, 1977. Cat. No. EM 44-10/1977., ISBN: 0-660-00917-7.

43. Collins, H.F. and D.G. Deaner. 1973. Sewage chlorination versus toxicity - a dilemma? J. Eng. Dev., 99: 761.

44. Bellania, M.A. and D.S. Bailey. 1975. A case history of some effects of chlorinated effluents on the aquatic ecosystem in the lower James River in Virginia. 45th Wat. Poll. Cont. Fed. Conf., Miami Beach.

45. Servizi, J.A. and D.W. Martens. 1974. Preliminary survey of toxicity of chlorinated sewage to sockeye and pink salmon. International Pacific Salmon Fisheries Commission, New Westminister, B.C., Progress Report, No. 30.

46. Tsai, C. 1973. Water quality and fish life below sewage outfalls. Trans. Am. Fish. Soc., 102: 281-292.

47. Blackwood, C.M. 1974. Fish processing capability of plants in the Atlantic Region. Paper Presented to the Federal-Provincial Atlantic Fisheries Committee Meeting on Utilization of Atlantic Marine Resources, Montreal.

48. Payne, J.F. and A. Rahimtula. 1981. Water chlorination as a source of aquatic environmental mutagens, p. 209-221. *In:* M.A.Q. Khan and R.H. Stanton (eds.), Toxicology of Halogenated Hydrocarbons: Health and Ecological Effects, Pergamon Press, New York, 396 p.

COAL AND COAL-BASED PRODUCTS

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INTRODUCTION

The late 1970s and early 1980s have seen a reemergence of coal as a basic, major energy source. In addition to fuel for solid fuel combustors and use in gasification plants, much consideration and effort is being put into the study of the feasibility of coal liquefaction for preparation of liquid fuels to replace those currently derived from petroleum. The Atlantic Region, especially Nova Scotia with its large coal reserves, will probably try to utilize more coal and coal-based products to reduce petroleum-based product use in the region.

Environmental problems may be associated with coal mining and use. Coal mining produces large quantities of solid and liquid wastes, as does coal washing and upgrading. Liquid wastes contain heavy metals leached from the coal and associated waste rock. Combustion of coal produces large amounts of air pollution and ash compared with oil. Coking and gasification of coal produces liquid and tarry by-products which, while serving often as a fuel or stock chemical, are a special concern since they have consistently been associated epidemiologically with cancer in man, and have been shown to induce cancers in experimental animals[1]. The carcinogenic activity of such materials has generally been ascribed to the presence of a class of chemicals known as polycyclic or polynuclear aromatic hydrocarbons (PAH) which are characterized as having three or more fused aromatic rings. Heteroaromatic PAH compounds are also well known since oxygen, nitrogen or sulfur can replace one or more carbon atoms in the rings. Furthermore, alkylation of various positions of the aromatic nucleus makes this class of compounds extremely complicated, and in most sources they occur as exceedingly complex mixtures. It is not unusual for modern analysts to identify many hundreds of different PAH compounds in a single natural or industrial source (e.g., Snook *et al.*[2]). PAH are present in fossil fuels, but the major environmental concern over these compounds is due to the fact that large amounts of non-alkylated PAH are formed during the combustion of all types of carbon-based fuels. Inefficient combustion, like that of wood in an open fireplace or a restricted air flow stove, produces many thousand times the amount of PAH produced during an equivalent amount of combustion in an efficient gas- or oil-fired furnace[3]. Liquid products from either coal liquefaction or coke production are extremely rich in PAH, with levels between 25 and 50 percent being common[4,5].

Coal-based tars and liquids contain substantial amounts of chemicals other than PAH such as the "tar acids" (phenols, naphthols, cresols, xylenols and others) and "tar bases" (pyridine, acridines, quinolines, etc.)

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as well as a large number of neutral sulfur, oxygen and nitrogen compounds (e.g., ethyl mercaptan, thiophene, diphenylene oxide, acetone, acetonitrile and carbazole). Chemicals such as phenolics are toxic to fish, taint their tissues and can induce avoidance behaviour.[6]. Discussion of the environmental role of these materials is beyond the scope of this article. This review will concentrate only on the marine environmental impact of coal mining, coal utilization and PAH resulting from the manufacture and use of coal-based products.

COAL MINING IN THE ATLANTIC REGION

Coal mining has a much greater impact on freshwater and terrestrial systems than it does on the marine environment due to the capacity of seawater to buffer acidic wastes. Atlantic Canada has an abundance of coal fields both inland and along the coast, some of which have been mined since the mid 1700's and early 1800's. In Nova Scotia, the Sydney Coalfield (total production 1863-1976 - 259 million tons) is by far the largest with the Pictou (63 million tons) and Springhill (35 million tons) coalfields being the next largest[7]. There are at least 12 coalfields in the Atlantic region with a large number of mines. Most of these discharge wastewater and runoff to the marine environment[8]. Associated with the mines are a number of coal preparation and washing plants where beneficiation of coal is carried out prior to its use. Figure 1 shows the location of discharge to the marine environment of waste from coal mining or beneficiation facilities in Atlantic Canada.

The impact of coal mining on the marine environment is attributable to: (1) the chemical nature of coal and coal-bearing rocks; and (2) the incidental production of quantities of fine particulate material in the mining, crushing or washing procedures. In these respects, the concerns about coal mining are similar to those about metal mining operations in the region. Also, all coal contains a significant quantity of sulphur (1.8% in some Nova Scotia coal to 6.1% in some New Brunswick coal) which contributes to a high level of acidity in mine wastewater which in turn increases the solubility of heavy metal ions. Some of the heavy metals included in Atlantic region coal are: chromium (3.0-65.4 ppm), copper (3.6-32.3 ppm), lead (1.8-55.9 ppm), titanium (510-2920 ppm), arsenic (4.3-311 ppm), and nickel (4.5-34.6 ppm).

Wastewater from coal mining operations in Canada is not well characterized since most studies have concentrated only on conventional parameters such as suspended solids, turbidity, BOD, pH, etc.[9]. Day *et al.*[8], however, in an environmental study of coal mining and processing facilities in Nova Scotia and New Brunswick did monitor heavy metals (Pb, Cu, Cd, Zn, Hg, Fe, Al, As, as well as cyanide) in addition to other environmental parameters such as suspended solids, alkalinity, conductivity, dissolved oxygen, temperature, etc. In this study, which examined 6 underground mines, 4 surface mines, 4 coal preparation plants, 2 abandoned mines, 1 restored surface mine and 1 major freshwater receiving body, seven were

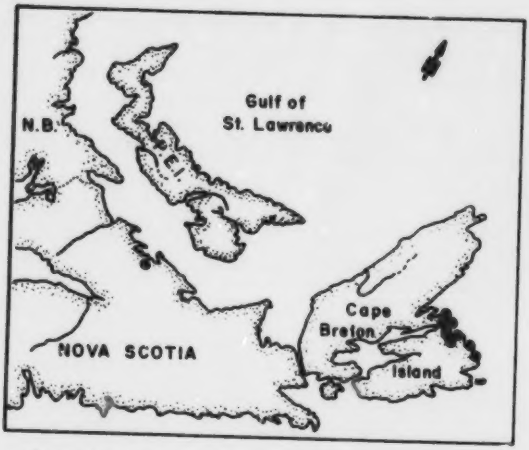

Figure 1: *Locations of discharge of coal mining and beneficiation wastes.*

found to be discharging effluents which were acutely toxic to fish (LT50 [rainbow trout] <96 hours). In these cases, and at most of the other sites examined by Day *et al.*[8], attempts to treat wastes were found to be inadequate. Fortunately the marine environment has the capacity to neutralize acidity and, thus, to reduce the toxicity of mine effluents. Therefore, the major impact of discharge is usually limited to the immediate vicinity of the outfall. In several instances large, highly coloured plumes have been observed extending well out to sea from outfalls, probably indicating the presence of a floc of hydrated oxides of iron. Precipitation and settling out of iron and other heavy metals could contribute to extensive contamination of marine sediments and biota in extended areas off such outfalls, however, no confirming data exist.

The principal noxious components of wastewater from the coal mining activities examined by Day *et al.*[8] were: acidity (low pH), heavy metals (particularly iron), and suspended solids. Metals (other than iron) in effluents were not considered of major concern with regard to toxicity. As mentioned earlier, however, even small quantities of trace elements can, over long periods, precipitate in the nearshore and become concentrated in sediments, generally contributing to a decrease in the quality of the nearshore environment.

COAL IN POWER GENERATION

Any expanded use of coal in North America is likely to affect the quality of the ocean through increased input of combustion products and various contaminants to the ocean by atmospheric transport. Atlantic

Canada is in the unfortunate geographic position of being downwind from the major part of industrial North America and thus receives a large proportion of air emissions from central Canada and the middle and eastern United States. Nova Scotia alone receives approximately 67,000 tonnes sulphur/y from these sources[10]. A large fraction of these emissions arise from coal combustion.In addition to SO_2 and NO_x, air emissions from the combustion of fossil fuels contribute large quantities of heavy metals and organic pollutants to the atmosphere in gaseous or particulate form which can be transported for hundreds of miles. Although acid rain is of little consequence to the well buffered oceans, its presence may indicate that other combustion products are reaching the marine environment, where they either go into solution or accumulate in marine sediments or in the biota.

MARINE IMPACTS FROM USE OF COAL

Air Emissions

The locations of all but one of the five coal fired electrical generating stations in Nova Scotia and New Brunswick are such that emissions are regularly transported over marine waters. It has been shown that the areas of greatest annual deposition (of SO_2 and other combustion products) are often within 5 to 30 kilometres of power stations, which means that most of the emissions from plants in the Region fall into the sea.

Increased concentrations of cadmium, nickel, zinc, copper and lead have been measured in soils and vegetation near coal-fired generating stations[12], and have been measured in aerial deposition within 10 km of a power station[13]. Measurement of the influx of these elements to the coastal marine environment from the atmosphere is difficult, however, because of the influence of mixing in the dynamic coastal system and the very diffuse nature of the influx. Nevertheless, it is estimated that atmospheric input of these contaminants is at least comparable with riverine input[14]. On an oceanwide basis, the significance of this input is difficult to assess. Although some of the elements are toxic, they may, as in the case of copper in the surface waters far offshore, be limiting elements essential for the biota in that environment.

Anthropogenic compounds such as organohalogens, hydrocarbons and other industrial organic materials do, on the other hand, present a threat to the marine environment. The combustion of coal contributes a number of organic compounds to the atmosphere. Most notable of these are the polynuclear aromatic hydrocarbons (PAH) which are transported in particulates or as volatiles over long distances, and which deposit in the marine environment, although their source might be well outside the Region[15]. The amount of PAH deposited in the marine environment from this source has not been determined, but the amount of PAH released from combustion can be estimated from the release of 1,200 metric tons of benzo[*a*]pyrene (BaP) per annum through combustion in the U.S.A.[16].

Since BaP, constitutes about 5-10% of the total PAH, Uthe[1] estimated total PAH release to be about 25,000 tonnes. The deposition is highest nearest to the release site and decreases rapidly with increasing distance from the source.

Liquid and Solid Wastes

The discharges (other than from cooling) of a power generating station usually account for less than 5% of the total water use, but contain the bulk of waterborne pollutants from coal. Water is used for ash disposal systems and equipment cleaning, and also arises from coal pile runoff, plant and yard drains, seepage from waste disposal areas, wastes from water treatment plants, and from boiler blowdown. Of all these, the largest wastewater stream at many stations is from direct release of the water used to transport coal ash from the station to a settling pond or lagoon. At coastal stations such as Lingan, Trenton, and Seaboard in Nova Scotia and Dalhousie in New Brunswick, this effluent enters the ocean or estuarine waters after leaving some sort of settling pond.

Ash disposal is a major problem at coal-fired generating stations. In Canada approximately 3.7 million tons of coal ash are produced per year. The amount of ash depends on the type of coal: bituminous yields 10%, sub-bituminous 15%, and lignite 30% ash. In Nova Scotia where some 1,940,000 tonnes of coal are burned annually, this represents at least 200,000 tons of ash annually, since Nova Scotia coal is bituminous and sub-bituminous with between 10-15% ash[12].

The environmental hazard from coal ash and its leachate arises from the large number of chemicals concentrated in the ash during combustion. These include most of the metals contained in the coal. Some, such as arsenic, barium, boron, cadmium, chromium, copper, selenium, and zinc, are discharged by leaching into the ash transport water. Leachate from abandoned ash disposal piles can be transported to the marine environment in runoff or groundwater. For example, a coal fly ash disposal basin and receiving water system were examined in South Carolina in which elevated levels of 10 toxic elements were found in sediments and biota[13].

The seriousness of the ash disposal problem is indicated by the fact that the Environmental Protection Agency in the U.S. considered classifying it as a "hazardous substance"[12]. It has been shown that the soluble portion of coal ash, containing various materials including molybdate, borate, fluoride, selenate and arsenate, is highly toxic to freshwater algae[17]. Although the freshwater environment is much more vulnerable than the ocean due to the vast dilution capacity and buffering capability of saltwater, ash transport water and leachate are potential marine contaminants.

Ash waste is not dumped in the ocean off the East coast of Canada but studies of ash and colliery waste dump sites off the coast of the U.K. indicate severe depletion of benthic fauna in localized areas, and potential migration of fine fly ash material in shallow water (<30 m) by means of wave action and tidal currents[18,19].

Large storage piles of coal at thermal generating plants and coal preparation facilities create coal pile runoff containing soluble elements from the coal. The volume of effluent from these sources is not large, but in most cases no special treatment is applied and the runoff from coal piles is combined with other runoff and directed to the receiving water directly or through settling ponds also used for ash slurry. The thermal plant at Dalhousie, New Brunswick, treats coal pile drainage along with other plant wastes, but, there is no treatment of such wastes at other facilities in the Atlantic region[20]. Heavy metal concentrations in leachate from coal piles are indicated in Table 1.

TABLE 1 - HEAVY METAL CONCENTRATIONS
IN LEACHATE FROM COAL PILES
AT THREE ATLANTIC REGION FACILITIES[20]

	Dalhousie Coal Pile Leachate[a] (mg/L)	Lingan Coal Pile Drainage (mg/L)	Victoria Jn. Leachate (mg / L)
Arsenic	40.0	0.70	0.1
Copper	8.0	0.52	0.005
Lead	0.2	<0.1	0.32
Nickel	7.8	1.4	-
Chronium	2.0	<0.05	-
Zinc	14.5	5	-
Mercury	-	-	0.0009
Aluminum	-	-	0.047
Cadmium	-	-	-

a - untreated leachate

Typical levels of trace organics in coal pile runoff are[20]: benzene, 2.6 mg / L; chloroform, 9.3 mg / L; methylene chloride, 22.0 mg L; toluene, 1.8 mg / L; and phthalate esters, 450 µg / L.

COAL-BASED PRODUCTS AND THEIR USE IN THE MARINE ENVIRONMENT

Coking and Coal Tar Production

Coking, the destructive distillation of coal to produce a high temperature fuel product, yields in addition to the coke and the gas a considerable amount of coal tar. Coal tar is a descriptive title for all liquids, tars and glassy solid condensates produced during the carbonization of coal. As a pyrolytic product, it is an extremely complex mixture of chemicals. Some coal tar is used directly, but a great deal of it is redistilled to yield a variety of fractions, two of which, creosote (green) oil and anthracene (yellow) oil contain very substantial proportions of PAH. The pot residue, known as coal tar pitch, also contains PAH, especially the higher boiling compounds. Coal tar pitch is used in protective coatings, fuels, paints, tarpaper, road surfaces, waterproofings and electrode production[22-27].

Coal is used extensively in the metallurgical industry both as a fuel and as raw material in the production of coke. The only coke oven facility in Atlantic Canada is located in Sydney, Nova Scotia and is associated with the Sydney Steel Corporation. In 1980 the Nova Scotia steel industry's demand for coal amounted to 570,000 tons, which was predicted to increase to 1,600,000 tons by the year 2000.

Coal-Based Liquids and the Marine Environment

The major widespread use of coal-based products in Atlantic Canada is undoubtedly the use of coal tar and creosote for the preservation of wood for use in both the terrestrial and the marine environments. Within the Atlantic region there are two plants which pressure treat wood: one at Truro, Nova Scotia, the other at Newcastle, New Brunswick. To be effective as a preservative in the marine environment, creosote must be at least 99% aromatic, since as little as 2% aliphatics allows attack by *Limnoria tripunctata* (ship worm)[28]. Creosote is enriched with respect to PAH as compared with coal tar. Creosote or creosote/coal tar mixtures are widely used for wood preservation in the marine environment[29-31] and, as such, represent a concentrated source of PAH to the local environment as freshly-placed pilings leach creosote for a number of years due to hydrative swelling[32]. Loss of creosote is greatest during the first years of use[33].

The coking of coal for use in iron and steel manufacture at Sydney, Nova Scotia, produces relatively large amounts of coal tar which enters the marine environment and results in elevated PAH levels. Lesser amounts of coal-based products enter the marine environment from other industrial applications. Coal tar and coal tar pitch are widely used in highway surfacings, including the residential and commercial use of coal tar suspensions as asphalt pavement sealers. The amount of PAH incorporated into or released from road surfaces into the marine environment is unknown.

Marine paints contain substantial amounts of PAH[28,29] and leaching of PAH from these paints may occur after application. Information on the amount of coal-based products used in marine paints in the region is not available.

PROBLEMS ASSOCIATED WITH COAL BASED PRODUCTS IN THE MARINE ENVIRONMENT

The current major environmental concern about coal-based products in the marine environment is due to their content of PAH. Metabolism of these compounds in most species, while leading to their excretion, also produces intermediates which are active carcinogens[34,35]. Thus we are faced with two potentially serious problems associated with PAH in the marine environment.

In the first case, questions are being raised regarding the occurrence of cancers and other sublethal effects in marine biota. Apparently, there are no reports identifying environmentally induced cancers in biota within the Atlantic region, however, tumors which are not confirmed as malignant have been found in groundfish[36]. Current investigative efforts in the region are not intensive, but information on the incidence of tumors and other pathobiological lesions is being collected in an internationally coordinated manner in the North Atlantic through the International Council for the Exploration of the Sea.

A second concern is over the human consumption of food species containing relatively high levels of PAH. These are cases in which the level of PAH in the environment is so high that even animals which metabolize them still contain certain PAHs[37]. Black *et al* studied the area around a creosote treatment plant and found PAH in all species studied, including white sucker *(Catostomus commersoni),* brown trout *(Salmo trutta)* and lamprey ammocoetes (presumable *Petromyzon marinus).* The lamprey ammocoetes, living and filter-feeding in the stream sediments had higher levels of PAH than the sediments in which they were living. It is not known if these animals possess mixed function oxidase capability and could thus metabolize PAH, although such capability has been demonstrated in Elasmobranchii[38]. Marine animals such as bivalves and certain crustaceans lack mixed function oxidase and accumulate substantial levels of PAH in their tissues, and it is in shellfish that the highest PAH levels are found [24,39,40].

Animals attached to or living near creosoted structures in the marine environment have been shown to contain elevated PAH levels in their tissues. Such levels, at least in mussel *(Mytilus edulis),* were found to decrease with increasing sampling distance from such structures[41]. Zitko[42] showed that the PAH pattern in mussels *(Mytilus edulis),* periwinkles *(Littorina littorea)* and whelks (various species) closely resembled that of creosote and suggested that the creosote-treated pilings within the collection area were the source of PAH contamination in these animals. Eaton and Zitko[43] found widespread PAH contamination in these species collected from a number of harbours in the Atlantic region where they were associated with creosoted structures.

Dunn and Fee[44] determined BaP in a wide variety of commercial seafoods. Samples of vertebrate fish did not contain detectable BaP, except for those products prepared in oils. Crab and shrimp samples contained little or no BaP. Levels in freshly caught lobster were uniformly low in tail meat, but specimens which had been taken from commercial storage in tidal pounds in which creosote-treated timbers had been used had levels of BaP as high as 281 ng/g wet weight with a range of 4-281 and a mean of 78.9

ng/g. Study of thirteen individual PAH compounds in lobster exposed to creosote showed that the total PAH concentration in the tail of the animal with 281 ng BaP/g was 4780 ng/g. Levels in the digestive gland were approximately ten times higher. Sirota and Uthe[45] confirmed the presence of PAH in lobster tail meat and digestive glands and also showed that oiled shellfish (mussels and lobster), sampled from areas known to have been contaminated by oil released by the breakup of the tanker "Kurdistan" in 1979, also contained substantial amounts of PAH.

The coke oven operation at the Sydney Steel Corporation in Sydney, Nova Scotia, has prompted an extensive survey of PAH sources in the Sydney area and PAH levels in sediment, biota, effluents and air. Lobster analyses carried out by DFO in 1980 and 1981 revealed that, at two sites in the harbour, PAH concentrations in lobster hepatopancreas were 78,100 ng/g and 21,000 ng/g wet weight (mean levels of 12 PAH compounds;10 animals per site). These levels are considerably in excess of the estimated background level in coastal lobsters, which is approximately 3,000 ng/g[46].

Results of the study sparked by the 1980-81 lobster surveys indicate widespread contamination of sediments in Sydney Harbour and identify the major source as the coke oven effluent. Total levels of the combined 12 PAH analyzed in sediment reached as high as 2829.06 mg/kg (dry weight) near the outfall and were still about 40-50 mg/kg as far as 8 km away toward the harbour mouth[47]. In response to this, lobster fishing in the contaminated area has been curtailed.

Even 16 km away within another arm of Sydney Harbour, sediment concentrations of 12 PAHs levels exceeded 7 mg/kg. BaP ranged from 108.53 mg/kg (dry) near the outfall in the South Arm to 0.37 mg/kg in the North Arm. At the mouth of the harbour, approximately 16 km from the outfall, levels of BaP had dropped to 0.026 mg/kg. This still exceeds what could be considered a "clean" background of about .005 mg/kg (total 12 PAH). Background levels in sediments from the Atlantic Abyssal Plain are as low as 0.018 mg/kg (dry weight) total PAH[48].

Sampling in Tokyo Bay, Japan, revealed high BaP levels (89,000 mg/kg dry weight) in sediments near the outfalls of two industries with coke ovens. The same situation is described in Osaka Bay, and in both cases water samples revealed the BaP to be associated with particulate matter[49].

Samples of sediment collected from the tar pond at Sydney Steel, which receives effluent from the steel mill and the coke ovens, show very high levels of all the 12 PAH measured. Notable are anthracene (44-1200 µg/g), benz[*a*]anthracene and chrysene (combined) (110-2700 mg/kg), BaP (180-1400 mg/kg), and phenanthrene (150-3100 mg/kg).

In summary, it can be concluded that the mining and use of coal itself as a fuel presents a limited and local threat to the marine environment. One exception to this conclusion is the long range aerial transport of coal combustion products to marine waters, which is a chronic and wide ranging problem not likely to be resolved in this century. The use of coal-based products, such as coal tar and creosote, presents a source of contaminants, particularly PAH, which poses a more serious threat to the marine environment, especially in coastal areas. The present dependence of marine interests on creosote as the only suitable preservative for marine timbers ensures that this source is unlikely to change for a long time. Further research is required to define the full nature of the threat of PAH to the marine environment, and to human consumers of marine produce.

REFERENCES

1. Uthe, J.F. 1979. The environmental occurrence and health aspects of polycyclic aromatic hydrocarbons. Can. Tech. Rept. Fish. Aquat. Sci., 914; iii + 30 p.

2. Snook, M.E., R.F. Severson, R.F. Arrendale, H.C. Higman and O.T. Chortyk. 1977. The identification of high molecular weight polynuclear aromatic hydrocarbons in a biologically active fraction of cigarette smoke condensate. Bietrage fur Tabakforschung, 9: 79-101.

3. Peters, J.A., D.G. Deangelis and T.W. Hughes. 1981. An environmental assessment of POM emissions from residential wood-fired stoves and fireplaces. *In:* Cooke, M. and A.J. Dennis (Eds.), Polynucleur Aromatic Hydrocarbons. Battelle Press, Columbus, Ohio. pp. 571-581.

4. Lao, R.C., R.S. Thomas and J.L. Monkman. 1975. Computerized gas chromatographic-mass spectrometric analysis of polycyclic aromatic hydrocarbons in environmental samples. J. Chrom., 112: 681-700.

5. Karr, C., W.T. Abel and J.R. Comberiati. 1977. Products from two-step coal liquefaction using three different first-step reactor packings. *In:* Ellington, R.T. (Ed.), Liquid Fuels from Coal. Academic Press, New York. pp. 19-44.

6. Sprague, J.B. and D.E. Drury. 1969. Avoidance reactions of salmonid fish to representative pollutants. Adv. Water Poll. Res., 5: 169-180.

7. Nova Scotia Department of Mines. 1978. A history of coal mining in Nova Scotia. Information Series No. 2.

8. Day, J.K., S.W. Dewis, B.J. Faught, D.A. Lord, H. Samant and G. Westlake. 1979. Environmental monitoring of coal mining and beneficiation facilities in Nova Scotia and New Brunswick, Canada. Env. Protection Service, Halifax, N.S. EPS-5-AR-79-5.

9. Dearborn Environmental Consulting Services. 1982. Identification and characterization of contaminants potentially present in Canadian coal wastewaters and assessment of the impact of these contaminants on the aquatic environment. For Water Pollution Control Directorate, EPS, Ottawa. March, 1982.

10. Wiltshire, J.F. 1979. The long range transport of atmospheric acids: a sulfur budget for Nova Scotia. Env. Prot. Serv. EPS-5-AR-79-8.

11. Ternan, G. Environmental Protection Service, Dartmouth, N.S., personal communication.

12. DOE, NH&W, EMR. 1980. Environmental health and safety aspects of coal-fired electrical power generation in Canada. A paper prepared jointly by Environment Canada, Health and Welfare Canada and Energy, Mines and Resources Canada. November, 1980.

13. Evans, D.W., J.G. Weiner and J.H. Horton. 1980. Trace element inputs from a coal burning power plant to adjacent terrestrial and aquatic environments. J. Air Poll. Control Assoc., 30(5): 567-573.

14. Bewers, J.M. 1979. Trace metals in the ocean. *In:* Long range transport of air pollution and its impacts on the Atlantic Region, workshop proceedings. Atmospheric Env. Service. Atl. Reg. October, 1979, pp. 64-65.

15. Windsor, J.G. and R.A. Hites. 1979. Polycyclic aromatic hydrocarbons in Gulf of Maine sediments and Nova Scotia soils. Geochimica et Cosmochimica Acta, 43: 27-33.

16. Suess, M. 1976. The environmental load and cycle of polycyclic aromatic hydrocarbons. The Science of the Total Environment, 6: 239-250.

17. Vocke, R.W., K.L. Sears, J. O'Tool and R. Wildman. 1980. Growth responses of selected freshwater algae to trace elements and scrubber ash slurry generated by coal-fired power plants. Water Res., 14: 141-150.

18. Eagle, R.A., P.A. Hardiman, M.G. Norton, R.S. Nunny and M.S. Rolfe. 1979. The field assessment of effects of dumping wastes at sea: 5. The disposal of solid wastes off the northeast coast of England. Fisheries Res. Tech. Rept., No. 51, ISSN 0308-5589. Lowestoft.

19. Nunny, R.S. 1978. A survey of the dispersal of colliery waste from Lynemouth Beach, Northumberland. Fish. Res. Tech. Rept., No. 43. ISSN 0308-5589, Lowestoft.

20. Dearborn Environmental Consulting Services. 1981. Characterization and treatability of drainage from coal piles at Canadian steam electric stations. For Water Pollution Control Directorate, EPS, Ottawa. March, 1981.

21. Plansearch. 1981. Chlorinated phenols/phthalate esters in industrial effluents. For Env. Prot. Service, Dartmouth, N.S. Plansearch, Lavalin. February, 1981.

22. Hey, D.M. (Ed.). 1966. Coal. *In:* Kingzett's Chemical Encyclopaedia. Bailliere, Tindall and Cassell, London. pp. 228-235.

23. McGannon, H.E. (Ed.). 1971. The Making, Shaping and Treating of Steel. 9th Edition, Herbig and Held, Pittsburg, P.A.

24. Larson, B.A. 1978. Occupational exposure to coal tar pitch volatiles at pipeline protective coating operations. Am. Ind. Hyg. Assoc. J., 39: 250-255.

25. Banfield, T.A. 1974. Paint technology in the marine environment. Paint Manufacture, 44: 18-19.

26. Brown, P. 1973. Coal tar-urethane and coal tar-epoxy finishes for metals. Metal Finishing, 71: 42-44.

27. The British Coal Carbonization Research Association. 1975. Review of coal tar technology: 25(2). Issued November, 1975. Chesterfield, Derbyshire, England.

28. Baechler, R.H. and L. Roth. 1961. Further data on the extraction of creosote from marine piles. Proc. Am. Wood Preservers' Assoc., 56: 124-129.

29. Mann, R.H. 1959. Introduction to "Symposium on economic implications and evaluations". *In:* Ray, D.L. (Ed.), Marine Boring and Fouling Organisms. University of Washington Press, Seattle. pp. 427-429.

30. Baechler, R.H. 1968. Further thought regarding variable performance of creosoted marine piling. Proc. Am. Wood Preservers' Assoc., 56: 177-126.

31. Graham, R.D. 1959. Preservation of Douglas Fir marine piling. *In:* Ray, D.L. (Ed.), Marine Boring and Fouling Organisms. University of Washington Press, Seattle. pp. 451-459.

32. Sweeney, T.R., T.R. Price, R.A. Saunders, S.M. Miller and F.G. Smith-Walton. 1958. Coal tar creosote studies, Part I. A method for the accelerated evaluation of marine wood preservatives. Proc. Am. Wood Preservers' Assoc., 14: 295-301.

33. Gibb, H. 1978. Coal tar, creosote and coal tar neutral oil. Federal Register, 43: 202. October 18, 1978.

34. National Academy of Science (U.S.A.). 1972. Particulate polycyclic organic matter. Washington, D.C.

35. Payne, J.F. 1977. Mixed function oxidases in marine organisms in relation to petroleum hydrocarbon metabolism and detection. Mar. Poll. Bull., 8: 112-116.

36. Morrison, C.M., R.G. Pippy, G. Shum, C. Annand and P.H. Odense. 1979. Histology and incidence of pseudobranch tumors in Atlantic cod *(Gadus morhua)* in Halifax Harbour. International Council for the Exploration of the Sea, Statutory Meetings, Warsaw, CM:1979/E:31.

37. Black, J.J., T.F. Hart, Jr., and E. Evans. 1981. HPLC studies of PAH pollution in a Michigan trout stream. *In:* Cooke, M. and A.J. Dennis (Eds.), Polynuclear Aromatic Hydrocarbons. Battelle Press, Columbus, Ohio. pp. 343-356.

38. Philpot, K.R.M., M.O. James and J.R. Bend. 1977. Metabolism of benzo(a)pyrene and other xenobiotics by microsomal mixed-function oxidases in marine species. *In:* Source, effects and sinks of hydrocarbons in the aquatic environment. Proceedings of the Symposium. Am. Inst. Biol. Sci., Arlington, Va., 1976. pp. 184-199.

39. Cahnman, H.J. and M. Kuratsume. 1957. Determination of polycyclic aromatic hydrocarbons in oysters collected in polluted water. Anal. Chem., 29: 1312-1317.

40. Zobell, C.E. 1971. Sources and biodegradation of carcinogenic hydrocarbons. Proceedings - Joint conference on prevention and control of oil spills. Am. Petroleum Institute, Washington, D.C. pp. 441-451.

41. Dunn, B.P. and H.F. Stich. 1975. Monitoring procedures for chemical carcinogens in coastal waters. J. Fish Res. Bd., Canada, 33: 2040-2046.

42. Zitko, V. 1975. Aromatic hydrocarbons in aquatic fauna. Bull. Environ. Contam. Toxicol., 14: 621-631.

43. Eaton, P. and V. Zitko. 1978. Polycyclic aromatic hydrocarbons in marine sediments and shellfish near creosoted wharf structures in eastern Canada. International Council for the Exploration of the Sea, 76th Statutory Meeting, Copenhagen, 1978. Document C.M. 1978/E:25.

44. Dunn, B.P. and J. Fee. 1979. Polycyclic aromatic hydrocarbon carcinogens in commercial seafoods. J. Fish. Res. Bd. Canada, 36: 1469-1476.

45. Sirota, G.R. and J.F. Uthe. 1981. Polynuclear aromatic hydrocarbon contamination in marine shellfish. *In:* Cooke, M. and A.J. Dennis (Eds.), Polynuclear Aromatic Hydrocarbons. Battelle Press, Columbus, Ohio. pp. 329-342.

46. Sirota, G.R., J.F. Uthe, A. Sreedharan, S. Matheson, C.J. Musial and K. Hamilton. 1982. Polynuclear aromatic hydrocarbons in American lobster *(Homarus americanus)* and sediments in the vicinity of a coking facility. *In:* Cooke, M. and A.J. Dennis (Eds.), Polynuclear Aromatic Hydrocarbons. Proceedings of the 7th International Symposium. Battelle Press, Columbus, Ohio. pp. 1123-1136.

47. EPS. 1982. Investigation of polycyclic aromatic hydrocarbon contamination of Sydney Harbour, Nova Scotia. An interim report. Env. Prot. Serv., Halifax. April, 1982.

48. Andelman, J.B. and M.J. Suess. 1970. Polynuclear aromatic hydrocarbons in the water environment. Bull. Wld. Hlth. Org., 43: 479-508.

49. Kawakami, Y. and H. Nishimura. 1976. Benzo(a)pyrene in sediment of Osaka Bay and Keihin Canal and its estimated sources. J. Oceanogr. Soc. Jap. 32: 175-181.

DISTRIBUTION AND BIOAVAILABILITY OF HEAVY METALS IN THE MARINE ENVIRONMENT

S. Ray* and J.M. Bewers**

INTRODUCTION

The term "heavy metal" is imprecise. One of the many definitions includes metals above atomic number 20, but excludes the alkali and alkali-earth metals, actinides and lanthanides. The heavy metals are natural constituents of the marine environment and normally occur at low concentrations. However, anthropogenic input may raise the level by orders of magnitude, especially in the estuarine and coastal areas. The group of metals with average concentrations less than 10^{-6} M constitutes the "trace metals". Several essential and non-essential metals fall in this group. Those like Cu, Zn, Fe, V, Cr, etc., though essential to living organisms may have detrimental effects when present in relatively elevated levels. The essential metals function in combination with organic molecules and are quite often metal-protein complexes. For example, the oxygen carriers hemoglobin and hemocyanin are metallo-proteins, as are many enzymes. Other metals like Hg, Cd, Pb, etc., do not have any role in life supporting systems and may be harmful even at low concentrations. Some metals, whether essential or non-essential, can inhibit enzyme activity by forming mercaptides with —SH groups and thus can be potentially toxic to living organisms.

Marine organisms can accumulate metals from their environment and can also normally adapt to natural fluctuations in the environment. The concerns with heavy metals in the marine environment are that high levels in commercial species can create problems for human consumption and that the organisms themselves may be affected.

A large part of the anthropogenic discharge to the water rapidly attaches to suspended particles and is deposited in sediments at concentrations many orders of magnitude greater than those in the overlying water. The analysis of water, suspended matter and sediment indicates total heavy metal load but generally gives a very unreliable picture of biological availability (bioavailability) of the metals. One of the reasons for this anomalous observation is that the determination of total metal concentration in the sample matrix does not take into account the presence of different chemical forms in which the metals may be present, each having its different biological availability. This is further complicated by the fact that various physicochemical factors like redox potential, pH, chelation, complexation, and biogeochemical transformations may all variously affect bioavailability of the metals to marine organisms.

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The level of metals in an organism is generally a reflection of the metals available to it in the environment. Consequently, the monitoring of heavy metal accumulation in the organism would provide the best indication of bioavailable heavy metal loading.

The heavy metals considered in this review have been restricted mainly, but not exclusively, to mercury and cadmium. They have been described as being most toxic to marine life[1] and are of current interest in the Atlantic Provinces.

DISTRIBUTION IN THE NATURAL ENVIRONMENT

Water

Some information exists about the incidence of most of the trace metals and metalloids in the waters of some areas of the western North Atlantic. There are data describing the more detailed geographic distribution of a few metals, particularly iron, manganese, copper, nickel, zinc and cadmium. This emphasis reflects interest in the marine geochemistry of trace metals and, to a lesser extent, limitations of sampling and analytical methods. Since 1975 there have been three reviews (all carried out as part of the work of the International Council for the Exploration of the Sea [ICES] of the distribution of trace metals in the North Atlantic[2-4]. The authors of the most recent of these reviews point out that the metals listed above have received more attention than, for example, mercury, lead and cobalt; other metals and metalloids, including chromium, arsenic, molybdenum, selenium and aluminum, have received little or no attention. Furthermore, the reviewers were unable to compare reliable data from different sources (analysts) because in no two cases were the procedures used for sampling, sample pretreatment (e.g., filtration, preservation) and analysis identical. Before turning to the discussion of trace metal distributions in the western North Atlantic, we shall discuss briefly recent developments in the field of metal determinations in seawater. This will clarify the difficulties and qualifications involved in the ensuing discussion.

Measurements of trace metal concentrations in seawater made prior to 1973 are generally considered unreliable due to a number of problems, including sample contamination during collection, inadequate storage and/or analysis. Even more recent data require careful evaluation of sampling, storage and analytical methodology before their validity is accepted. During the last five years especially, some sampling and analytical techniques have been found to yield data that are "oceanographically consistent" for several trace metals. These developments have revolutionized our understanding of the marine geochemistry of trace metals and have led to a better appreciation of the processes by which metals are transported, mixed and sedimented in the ocean. A popular misconception is that the concentrations of trace metals will always be substantially higher in nearshore waters than offshore. The fallacy of this view, in the cases of several metals, has been adequately demonstrated by the recent data. Furthermore, the general decline in reported concentrations of metals in the ocean that has occurred during the last few decades and which reflects improved sampling and analytical methods, has now stopped in the cases of the most commonly determined metals. Thus, we are more confident now that the levels of the metals currently reported in the ocean are accurate and will not be significantly revised in the future. Of course, a price has been paid for this rapid advance: most of the "old" data have been discarded. This, in turn, has reduced the size of our data base. Nevertheless, the oceanographic consistency of the newer data, and the consequently better understanding of the distribution of trace metals in the ocean, enables us to extrapolate from the small body of reliable measurements to larger regions of the ocean such as the western North Atlantic.

The distribution of trace metals in the deep ocean beyond the continental shelf will be discussed first. Table 1 provides information on the concentrations and vertical distributions of a range of metals and metalloids in the western basin of the North Atlantic Ocean. This table includes data derived from direct measurements in this area, and measurements from other parts of the ocean that probably apply to the western basin (the areas from which these latter data are derived are given). The table also describes profiles of vertical distribution found for each metal in pelagic areas; these

TABLE 1 - CONCENTRATIONS AND VERTICAL DISTRIBUTIONS OF METALS IN THE PELAGIC NORTHWEST ATLANTIC

Metal	Conc. ng/l	Depth Range	Data Source	Profile Type (see Fig. 1)	Upper Surface Layer	Near-Bottom Layer
Cd	40	Mid	NWA[e]	Nutrient-P	Increases	Nil
Cu	110	Mid	NWA	Constant	Increases	Increases[a]
Ni	230	Mid	NWA	Nutrient-P	Nil	Nil
Zn	350	Mid	NWA	Nutrient-Si	Increases	Variable
Fe	450	Mid	NWA	Input / Scav.	Increases	Variable[b,c]
Mn	30	Mid	NWA	Input / Scav.	Increases	Variable[b,c]
Pb	3	Mid	NA	Input / Scav.	Increases	Nil
Hg	1	Mid	NA	Constant	Increases	Nil[c]
Mo	10000	Mid	NA	Constant	Nil	Nil
Al	1000	Mid	NWA	Input / Scav.	Increases	Increases
Cr	230	Mid	NWA	Nutrient-P	Nil	Nil
As	1500	Mid	-	Constant	Decreases[d]	-
Se	150	Mid	NA	Nutrient-P	-	-

[a] In regions of non-calcareous sediments.

[b] Can show both, losses due to scavenging and slight increases due to releases from sediments.

[c] Increases found in areas of tectonic activity.

[d] Very slight decreases in surface layer.

[e] NA: North Atlantic; NWA: Northwest Atlantic

profiles are illustrated in Figure 1. Metals which recycle rapidly in the upper layers of the ocean (because of uptake by marine organisms and subsequent dissolution or regeneration as the biogenic detritus sinks through the water column) have profiles in the upper portion of the ocean similar to those of the major nutrients silicate, nitrate and phosphate (Fig. 1a). These profiles show depletion at the surface, gradually increasing concentrations with depth to a maximum at the depth of maximum nutrient concentration and (in the cases of metals that show profiles similar to phosphate or nitrate) gradual declines in concentration with depth. Other metal concentrations

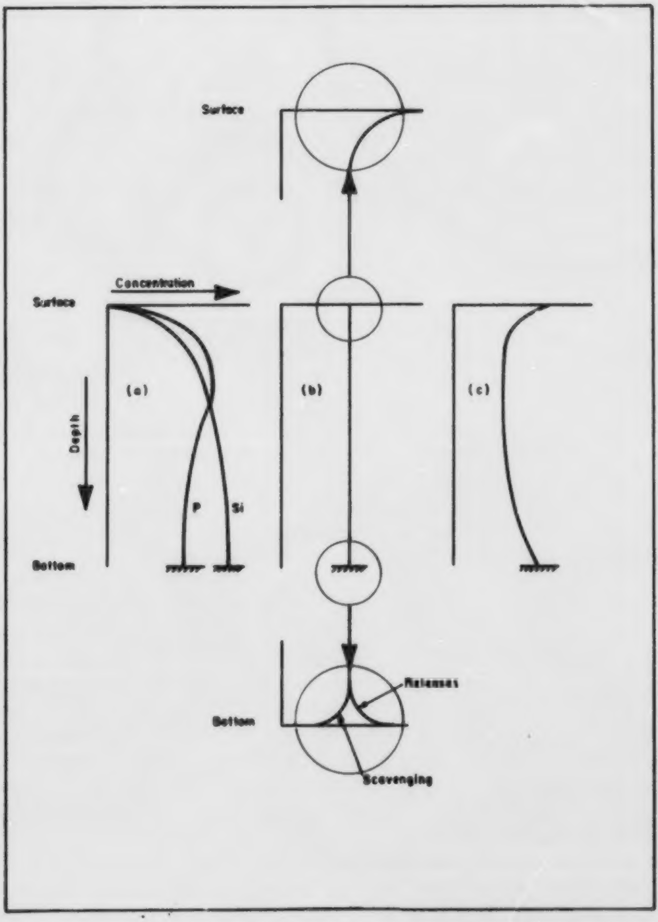

Figure 1: Various types of vertical metal profile in the deep ocean.

may not vary with depth, implying little biological or chemical control of their distribution (Fig. 1b). Profiles which show high concentrations at the surface and decline with depth to some constant concentration indicate high influxes at the surface and scavenging from solution in the water column (Fig. 1c). Two other factors may influence the distribution of some metals. The first of these is the rate of supply of metals to the ocean surface, particularly through atmospheric deposition. This is obviously important to metals having profiles like that shown in Fig. 1c, but it may also cause a marked enrichment of otherwise depleted metals near the surface of the ocean. The second factor is the release of dissolved metals from pelagic sediments or from volcanic activity on the sea floor, or, alternatively, the selective scavenging of metals from solution near the sediment/water interface by particles in the so-called "bottom nepheloid layer". (This layer usually extends a few tens or hundreds of meters above the bottom and contains higher concentrations of suspended particles than the overlying intermediate water.) Processes occurring close to the sea bottom may therefore control the vertical profiles of certain metals.

We next discuss those factors which control the concentration and distribution of metals in inshore areas. The first of these factors is the effect of mixing between freshwater runoff from land and seawater. Most such mixing occurs very close to shore except in large semi-enclosed embayments with limited exchange with the open ocean, such as the Gulf of St. Lawrence. The composition of river water is different from seawater and many metals have higher concentrations in river water than in the ocean[5]. Thus, as salinity decreases closer to the shoreline the concentrations of many metals in the water column increase. Only a relatively few metals and metalloids have lower concentrations in rivers than in the open ocean, so that this inverse relationship with salinity is quite common. However, since most mixing occurs very close to shore, increased concentrations due to mixing are confined to the nearshore. Even where the mixing extends a considerable distance offshore, as in the Gulf of St. Lawrence, the salinity in such areas is fairly close to that offshore and metal concentration increases are relatively small. If a metal is conservative in its behaviour, i.e., unaffected by biological or chemical processes, its nearshore concentration will reflect mixing and will lie on the linear relationship between its freshwater and seawater concentrations. Such a metal is molybdenum. However, most metals are affected by biological and chemical processes, and behave non-conservatively: their mass in the water column of the mixing zone is not conserved and they are either lost by removal to the sediments or are introduced to the water column by release from the sediments. An extreme example of removal in this way is iron which is precipitated from within mixing regimes. Initially the dissolved iron precipitates onto particles that are suspended in the water column, and as these particles settle to the bottom the precipitated iron is removed to the sediments. Manganese behaves similarly but it is also released from bottom sediments at the nearshore so that its concentration in nearshore water, such as estuaries, may be either depleted or augmented with respect to conservative mixing. The features of conservative and non-conservative mixing are shown in Figure 2.

Several other factors influence metal concentrations in the nearshore, the most important being processes of sediment transport and resuspension which cause high turbidity in coastal waters. Metals, like iron, that are rapidly exchanged between particles and water can show marked variations in concentration, both in time and space, that result from sediment resuspension. Another factor is related to the different sources of trace metals to coastal regions: different freshwater flows into a particular

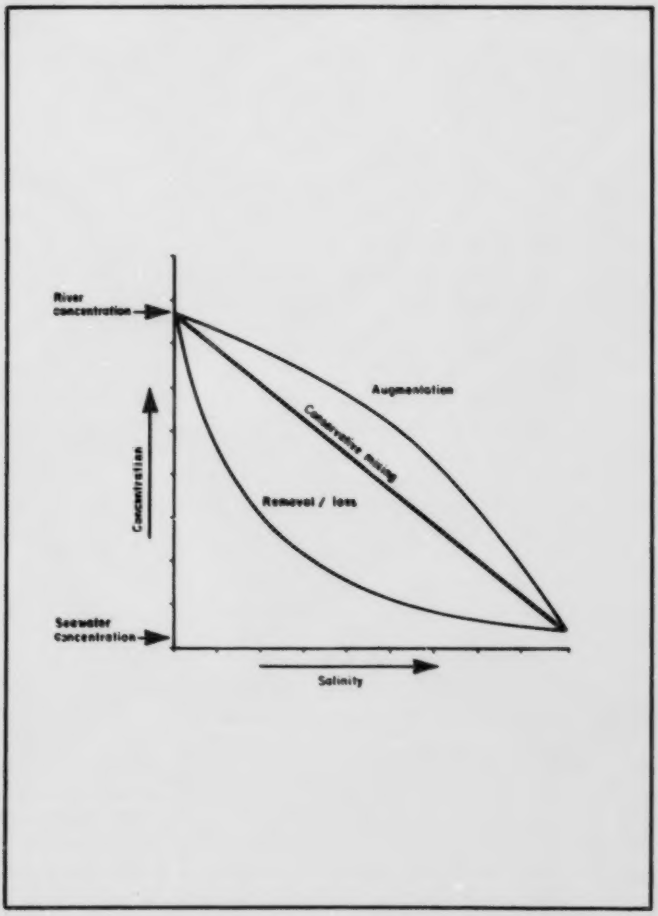
River concentration
Augmentation
Conservative mixing
Concentration
Removal / loss
Seawater Concentration
Salinity

Figure 2: *Different mixing relationships in estuaries.*

coastal area may differ in composition. Furthermore, metals are also introduced through atmospheric deposition (rain, snow, dust.). All these influxes cause complex mixing sequences more difficult to predict than straightforward mixing between two parcels of water of different compositions. Finally, the effects of contamination of the ocean by the products of human and industrial activities are most noticeable in the nearshore areas. Rapid dilution and dispersion of such contaminants produces very low concentrations of contaminants from point sources, both in the coastal zone and in the open ocean. In the case of metals, it has been shown that very severe contamination of the St. Lawrence River would not make discernable differences to the metal concentrations outside the St. Lawrence estuary[6]. Only those metals which are mobilized by man's activities on a global scale in amounts well above those moving through the environment naturally are likely to cause large-scale contamination of the ocean. Lead and possibly cadmium are examples of such metals.

Table 2 provides information relevant to the concentrations of metals and metalloids in coastal waters, and also summarizes some direct measurements of metal concentrations in the St. Lawrence estuary[7]. It should be stressed that our present understanding of the mixing relationships for most metals is very poor. This results from the complexity of estuaries, within which most nearshore mixing occurs, and reflects the difficulty of distinguishing effects due to mixing from the influences of other processes.

In summary, the concentrations of metals throughout the marine waters of the western North Atlantic, in the Gulf of St. Lawrence and in coastal Atlantic areas do not vary widely. Regions of localized metal contamination such as in Belledune Harbour and Chaleur Bay exist, but these are not extensive and have no profound effects on the greater marine environment. The major contamination of the whole ocean, including the western North Atlantic, is confined to a few metals, particularly lead and cadmium. In most areas it is possible to predict with reasonable confidence the prevailing levels of metals in the water column. The only areas in which this is difficult are those close to the boundaries of the ocean, i.e., near to shore, near to the sediments and very close to the surface.

Sediment

Sediment analysis can be particularly helpful in detecting pollution sources and selecting critical sites for routine analysis for contaminants. Most contaminants do not remain solubilized in the water column but are rapidly adsorbed by particulate matter and eventually settle to the bottom, thus escaping detection by water analysis. However, such metals do not necessarily remain "sediment-bound" for all time and may be recycled biogeochemically both within sediment components and back to the water column. The extent of recycling is significant, since "sediment-bound" metal may become bioavailable. Vertical profiles of metals in sediments are very helpful in providing the history of pollution over an extended period

TABLE 2 - INFORMATION RELEVANT TO THE CONCENTRATIONS OF METALS IN ESTUARINE AND NEARSHORE AREAS

Metal	Riverine Conc. ng/L	Seawater Conc. ng/L	Central St. Lawrence Estuary Conc.[c] ng/L	Mixing Behaviour	Releases from Sediments?	Common Contaminant?
Cd	110[a]	40	93	?	Possibly	Yes
Cu	2700[a]	110	840	Conservative?	Possibly	No
Ni	1800[a]	230	520	Conservative	No	No
Zn	10000[a]	350	2110	Conservative?	Possibly	Yes
Fe	68000[a]	450	5500	Removal / Loss	Possibly	No
Mn	7700[a]	30	2420	Removal / Loss	Yes	No
Pb	1000	3	-	Removal / Loss?	No	Yes
Hg	20	1	-	?	No	Yes
Mo	1000	10000	-	Conservative	-	No
Al	93000[a]	1000	-	-	No	No
Cr	700[b]	230	-	Removal / Loss	-	-
As	2000	1500	-	-	-	No
Se	200	150	-	Conservative?	-	No

[a] After Reference 5.

[b] Measured in the St. Lawrence.

[c] Reference 7.

and the background (baseline) levels for comparison. The difference between the two measured levels, i.e., the sediment and the background levels, may give an estimate of "extent of pollution".

Comparison of heavy metal pollution of sediments is often difficult because there are no standards for collection, storage, analysis and reporting of results. Samples must be kept in cold storage to prevent chemical transformation by biological activity. The particle size of the sediment to be analyzed must be determined, since metal content of the sediments decreases as particle size increases. Normally the fraction less than 63 µm is most important because: (a) - trace metals are present mainly in this fraction, which includes clay and silt; and (b) - this material is nearly equivalent to the material carried in suspension (the most important mode of transport of sediment).

Estimates of total metal concentration in the sediment may be insufficient to assess the impact of contaminated sediment on the environment since only a part may be bioavailable. Therefore, sequential chemical leaching procedures have been devised to define the distribution of metals within various components of the sediment. However, the lack of a standardized extraction scheme makes it difficult to compare results of analysis.

Chemical analysis of sediment for heavy metals usually involves digestion of the sediment with strong acids and determination of the extracted metals by atomic absorption spectroscopy. Complete digestion is normally performed with hydrofluoric acid combined with perchloric acid, sulphuric acid or nitric acid, in a Teflon bomb. Equally reliable estimates have been obtained by "strong acid" leaching by *aqua regia*, inverse *aqua regia* or some other combinations of strong acids. Complete dissolution of all components of sediments may be unnecessary, since the pollution effects are usually associated with surface components of the sediments.

The contamination of sediments by lead, zinc, and cadmium has been reviewed[8]. Loring[9,12] described metal concentrations in several east coast sediments including the Gulf of St. Lawrence, Bay of Fundy and the contaminated Belledune Harbour. The distribution of Cu, Zn, Cd, and Pb levels in estuarine surficial sediments of Miramichi, Nepisiquit and Restigouche Rivers has also been studied[13,14]. Average concentrations of several heavy metals in marine sediments from these and other studies and comparative values from other geographical locations are given in Table 3.

Biota

Various marine organisms can accumulate metals from their surroundings. Extensive data on heavy metal levels in marine biota were compiled by Vinogradov[24]. Davis[25] has compiled recent values of heavy metal levels in marine zooplankton. Freeman *et al.*[26] reported total mercury levels in a large number of fish and shellfish from the Canadian Atlantic coast. Methylmercury levels in fish from New Brunswick and Nova Scotia

TABLE 3 - AVERAGE METAL CONTENTS (µg/g dry weight) OF SEDIMENTS FROM VARIOUS GEOGRAPHIC LOCATIONS

Location	Cu	Zn	Pb	Cd	Mo	Ni	As	Hg	Reference
Marine Clays	25	165	80	0.21	8	225	13	0.08	15
Mean Sediment	33	95	19	0.17	2	52	8	0.19	15
Saint John Inner Harbour	16	53	24	0.16	3	16	-	0.04	16
Saint John Outer Harbour	15	65	-	0.07	-	-	-	0.03	16
Bay of Fundy	15	51	20	0.22	-	15	9	0.03	10
Miramichi Estuary	27	219	83	-	-	45	-	-	14
Upper St. Lawrence Estuary	36	185	34	0.26	-	27	6	0.38	18, 19
Lower St. Lawrence Estuary	24	115	30	0.22	-	33	7	0.45	18, 19
Open Gulf of St. Lawrence	25	84	21	0.26	-	36	6	0.22	18, 19
Saguenay Fjord	27	130	48	0.25	-	28	21	3.66	10, 20
Placentia Bay, Nfld.	34	80	32	-	-	41	-		17
Nepisiguit Estuary	36	248	21	1.5	-	-	-	-	13
Restigouche Estuary	9	50	13	0.74	-	-	-	-	13
Belledune Harbour	-	1710	-	32	-	-	-	-	12
Narragansett Bay, R.I.	53	110	44	0.47	-	18	-	-	21
Severn Estuary (U.K.)	38	280	119	-	-	36	-	-	22
Dredge spoils (U.K.)									
(a) Thames Estuary	46	220	93	1.8	-	50	-	0.78	23
(b) Mersey Docks	115	490	198	1.9	-	48	-	3.20	23

Banks were reported by Zitko *et al.*[27]. Dadswell[28] reported on mercury levels in striped bass from the Saint John River, New Brunswick, while Ray *et al.*[29] studied the levels in fish from the Annapolis and Shubenacadie Rivers in Nova Scotia. Cadmium levels in hepatopancreas of lobsters from various locations in Atlantic Canada were studied by Uthe and Freeman[30]. Ray *et al.*[31] studied the distribution levels of cadmium in an extensive variety of marine biota collected from the vicinity of Belledune Harbour, New Brunswick. Cossa and Bourget[32] have presented data on cadmium and lead levels in mussels, *Mytilus edulis,* from the St. Lawrence estuary and the northwestern Gulf of St. Lawrence. Arsenic is of considerable interest, but is rarely included in studies of metal uptake by marine organisms probably because arsenic analysis is difficult and sample preparation is also cumbersome.

Comprehensive reviews on mercury[33] and cadmium[34] in the Atlantic Provinces have been published. Heavy metal contents of contaminated and uncontaminated water, sediment, suspended matter, and plankton tissues from various areas around the world are reviewed annually in the literature issue of the "Journal of Water Pollution Control Federation", which is of immense value to people working on heavy metal pollution.

ACCUMULATION FROM DIFFERENT SOURCES

Water

Accumulation from water takes place by passive diffusion through the body surface or from water passing through the gills and subsequently through the bodies. Metals taken up in this way may be translocated in the body tissues. Rates of uptake of several metals (copper, silver, zinc, cadmium, mercury, arsenic and manganese) were found to increase in the polychaete *Nereis diversicolor* with increase in the concentration[35]. The same relationship was also observed for cadmium in the polychaete *Nereis virens*[36], shrimps *Pandalus montagui*[37] and *Lysmata seticaudata*[38], bivalves *Mytilus edulis*[38] and *Macoma balthica*[39], and for lead in the oyster *Crassostrea virginica*[40].

Some organisms, like bivalves, can temporarily prevent absorption of trace metals by closing their shells but, as yet, there is no evidence that any animal can altogether prevent the entry of metals into its body. Factors like physiological conditions, temperature, salinity, animal size, presence of other metals, and complexing agents, etc., are known to affect bioaccumulation.

Heavy metals are found in a number of different chemical forms in the environment. The relative proportion of each form will be an important factor governing its biological availability. George and Coombs[41] found that prior complexation of ionic cadmium with EDTA, humic acid, and alginic acid doubled the final tissue concentration of cadmium in *Mytilus edulis* compared to the level in animals exposed to ionic cadmium. However, in a similar study, Ray *et al.*[42] observed a significant reduction in accumulated cadmium in the polychaete *Nereis virens* and the crustacean *Pandalus montagui* exposed to cadmium EDTA. Rainbow *et al.*[43] have also observed a significant decrease in cadmium uptake by the barnacle *Semibalanus balanoides* exposed to humate, alginate, and EDTA complexes of cadmium.

Competition between chemically similar ions can also affect uptake of metals. The uptake of cadmium by the polychaetes *Nereis virens*[42] and *Nereis diversicolor*[44] was significantly reduced in the presence of zinc. Similar effects were observed when the crustacean *Pandalus montagui*[37] was exposed to cadmium in the presence of zinc. A progressive reduction of cadmium uptake in the presence of an increasing amount of zinc was also observed.

The concentration of a heavy metal within an organism can vary considerably with size. The extensive work of Boyden[45] described size related variations in heavy metal levels in molluscs. Size related variations in the concentration of lead were observed in mussels[46] and cadmium in polychaetes[36].

Sediment

Heavy metal concentration in sediments, especially in coastal and estuarine areas, is normally several orders of magnitude higher than in the overlying waters. However, despite the elevated concentrations of the metals in the sediments, their role as a source of metals to benthic and bottom feeding organisms is still uncertain.

Animals from areas with contaminated sediments have, in some cases, elevated metal levels[47-49]. Goldberg *et al.*[50] have suggested that bivalve molluscs may be suitable bioindicators of pollution because of their sessile nature and high bioaccumulation capacity for several pollutants. Mussels of the genus *Mytilus* have been used extensively for this role since they are widely distributed in both hemispheres at different temperature regimes. On the other hand, in an extensive laboratory study, Neff *et al.*[51] found that out of 136 combinations (3 sediments, 5 invertebrates, 8 heavy metals and 3 salinities) only 36% of the combinations showed statistically significant relationships between heavy metal concentrations in the sediment and in the tissues of the exposed animals.

A number of factors control the uptake of heavy metals from marine sediments by deposit feeding and benthic organisms. Biogeochemical factors, sediment texture and composition, salinity, and microbial activity can all affect bioavailability of the metals in the sediment to the organism. Physical disturbances like turbulence can promote metal exchange between sediments, particulate matter, and water, and cause redistribution of the elements. Ray and McLeese[52] and Bryan[53] have reviewed the processes affecting biological availability of heavy metals in marine sediments. The consensus is that bioavailability depends upon the amount of metals leached into the surrounding waters.

It is being gradually recognized that composition, texture, and size of sediment have as important a role as metal concentration in determining bioavailability of the metals present in sediments. Increased sediment organic level was found to reduce availability of silver and lead to *Nereis diversicolor* and zinc to *Scrobicularia plana*[54]. Langston[55] observed that mercury levels in *Scrobicularia plana* increased with lower organic levels in the estuarine sediment and decreased at high levels. Cooke *et al.*[56] observed that cadmium bound to biogenic calcium carbonate is readily available to the cockle *Cerastoderma edule,* but that metal bound to precipitated calcium carbonate is much less bioavailable. Furthermore, cadmium bound to iron and/or manganese oxides is not bioavailable. Luoma and Bryan[57] observed that the availability of lead to *Scrobicularia plana* is controlled by the concentration of iron in the sediment and can be predicted from the lead to iron ratio in an acid extract of surface sediments.

Food

Experimental work with plaice (*Pleuronectes platessa* L.) proved that food can be a major source of heavy metals like mercury and cadmium[58,59]. Studies with euphausiids *(Meganyctiphanes norvegica)*[60] and lobsters *(Homarus americanus)*[61] have also proved that food can be a major source for cadmium in animal tissues.

CONCLUSIONS

The levels of metals in marine species in the western North Atlantic have generally not been elevated by man's activities. Exceptions occur for some metals on a local scale in polluted harbours. If the level of heavy metals in an organism is assumed to be an indicator of heavy metals available to it in the environment, then the analysis of organisms and the exposure matrix should provide a reliable way of monitoring biologically available heavy metal. Since uptake of metals varies with species, it is necessary to select a single species for monitoring bioavailability. Much of what is known about bioavailability and bioaccumulation of heavy metals by marine organisms has come from laboratory studies. However, there are inherent dangers in trying to extrapolate the results to field situations. In the case of sediments, knowledge of the various forms in which the metal exists can be gained by chemical extraction techniques. Metal speciation in interstitial and sea water will aid in a better understanding of the process.

Since the speciation of metals in the field environment is largely unknown, it will still be difficult to relate laboratory findings to the field situation. Conducting field experiments may be the ultimate way to determine bioavailability in real life situations.

REFERENCES

1. Ketchum, B.H., V. Zitko and D. Saward. 1975. Aspects of heavy metals and organohalogen pollution in aquatic ecosystems, p. 75-90. *In:* A.D. McIntyre and C.F. Mills (eds.), Ecological Toxicology Research, Plenum Publishing Corp., New York.

2. Jones, P.G.W. 1975. Review of the distribution of selected trace metals in the waters of the North Atlantic. *In:* Report of the Advisory Committee on Marine Pollution, 1975, p. 16-33, International Council for the Exploration of the Sea, Cooperative Research Report No. 50.

3. Bewers, J.M. 1979. Trace metals in waters within the jurisdictional area of the International Commission for the Northwest Atlantic Fisheries. Bedford Institute of Oceanography, Report No. BI-R-79-2.

4. Topping, G., J.M. Bewers and P.G.W. Jones. 1980. A review of the past and present measurements of selected trace metals in sea water in the Oslo Commission and ICNAF/NAFO areas. International Council for the Exploration of the Sea, Cooperative Research Report No. 97, 43 p.

5. Yeats, P.A. and J.M. Bewers. 1982. Discharge of trace metals from the St. Lawrence River. Can. J. Earth Sci. 19: 982-992.

6. Yeats, P.A., J.M. Bewers and A. Walton. 1978. Sensitivity of coastal waters to anthropogenic trace metals emission. Mar. Poll. Bull., 9: 264-268.

7. Bewers, J.M. and P.A. Yeats. 1978. Trace metals in the waters of a partially mixed estuary. Estuarine and Coastal Marine Science, 7: 147-162.

8. Warren, L.J. 1981. Contamination of sediments by lead, zinc and cadmium: A review. Environ. Pollut. (Ser. B), 2: 401-436.

9. Loring, D.H. 1975. Mercury in the sediments of the Gulf of St. Lawrence. Can. J. Earth Sci., 12: 1219-1237.

10. Loring, D.H. 1979. Baseline levels of transition and heavy metals in the bottom sediments of the Bay of Fundy. Proc. N.S. Inst. Sci., 29: 335-346.

11. Loring, D.H. 1981. Potential bioavailability of metals in eastern Canadian estuarine and coastal sediments. Rapp. P-V Reun. Cons. Int. Explor. Mer., 181: 93-101.

12. Loring, D.H., J.M. Bewers, G. Seibert and K. Kranck. 1980. A preliminary survey of circulation and heavy metal contamination in Belledune Harbour and adjacent areas, p. 35-47. *In:* Cadmium Pollution of Belledune Harbour, New Brunswick, Canada (J.F. Uthe and V. Zitko, eds.), Can. Tech. Rep. Fish. Aquat. Sci., 963, v + 107 p.

13. Ray, S. and W. White. 1977. Some observations on heavy metal concentration in northeastern New Brunswick estuarine surficial sediments. Fish. Mar. Serv., Res. Dev. Tech. Rep. 696, 23 p.

14. Willey, J.D. and R.A. Fitzgerald. 1980. Trace metal geochemistry in sediments from Miramichi estuary, New Brunswick. Can. J. Earth Sci., 17: 254-265.

15. Bowen, H.J.M. 1979. Environmental Chemistry of the Elements. Academic Press, New York.

16. Ray, S. and S.D. MacKnight. 1984. Trace metal distributions in Saint John Harbour sediments. Mar. Poll. Bull. (in press).

17. Willey, J.D. 1976. Geochemistry and environmental implications of the surficial sediments in northern Placentia Bay, Newfoundland. Can. J. Earth Sci., 13: 1393-1410.

18. Loring, D.H. 1978. Geochemistry of zinc, copper and lead in the sediments of the estuary and Gulf of St. Lawrence. Can J. Earth Sci., 15: 757-772.

19. Loring, D.H. 1979. Geochemistry of cobalt, nickel, chromium and vanadium in the sediments of the estuary and open Gulf of St. Lawrence. Can. J. Earth Sci., 16: 1196-1209.

20. Loring, D.H. 1976. The distribution and partition of zinc, copper and lead in the sediments of Saguenay Fjord. Can. J. Earth Sci., 13: 960-971.

21. Eisler, R., R.L. Lapan, Jr., G. Telek, E.W. Davey, A.E. Soper and M. Barry. 1977. Survey of metals in sediments near Quonset Point, Rhode Island. Mar. Pollut. Bull., 8: 260-264.

22. Chester, R. and J.H. Stoner. 1975. Trace elements in sediments from the lower Severn estuary and Bristol Channel. Mar. Pollut. Bull., 6: 92-95.

23. Murray, L.A. and M.G. Norton. 1979. The composition of dredged spoils dumped at sea from England and Wales. Fish. Res. Tech. Rep. No. 52, MAFF Direct. Fish. Res., Lowestoft, U.K., 10 p.

24. Vinogradov, A.P. 1953. Elementary chemical composition of marine organisms. Sears Foundation of Marine Research, Yale University Press, New Haven, Conn.

25. Davies, A.G. 1978. Pollution studies with marine plankton, Part II, Heavy metals. Adv. Mar. Biol., 15: 381-508.

26. Freeman, H.C., D.A. Horne, B. Mctague and M. McMenemy. 1972. Mercury in some Canadian Atlantic coast fish and shellfish. J. Fish. Res. Board Can., 31: 369-372.

27. Zitko, V., B.J. Finlayson, D.J. Wildish, J.M. Anderson and A.C. Kohler. 1971. Methylmercury in freshwater and marine fishes in New Brunswick, in the Bay of Fundy and on the Nova Scotia banks. J. Fish. Res. Board Can., 28: 1285-1291.

28. Dadswell, M.J. 1975. Mercury, DDT, and PCB content of certain fishes from the Saint John estuary, New Brunswick. Trans. Can. Soc. Environ. Biol., 1975 Annual Meeting, p. 133-146.

29. Ray, S., B.M. Jessop, J. Coffin and D.A. Swetnam. 1984. Mercury and polychlorinated biphenyls in striped bass *(Morone saxatilis)* from two Nova Scotia rivers. Water, Air and Soil Pollut. (in press).

30. Uthe, J.F. and H.C. Freeman. 1980. Cadmium in hepatopancreas of American lobster *(Homarus americanus)* from eastern Canada, p. 73-76. *In:* J.F. Uthe and V. Zitko (eds.), Cadmium Pollution of Belledune Harbour, New Brunswick, Canada, Can. Tech. Fish. Aquat. Sci., 963.

31. Ray, S., D.W. McLeese, C.D. Metcalfe, L.E. Burridge and B.A. Waiwood. 1980. Distribution of cadmium in marine biota in the vicinity of Belledune, p. 11-34. *In:* J.F. Uthe and V. Zitko (eds.), Cadmium Pollution of Belledune Harbour, New Brunswick, Canada, Can. Tech. Rep. Fish. Aquat. Sci., 963, v + 107 p.

32. Cossa, D. and E. Bourget. 1980. Trace elements in *Mytilus edulis* L. from the estuary and Gulf of St. Lawrence, Canada: Lead and cadmium concentrations. Environ. Pollut. (Ser. A), 23: 1-8.

33. Wilson, R.C.H. and I.C. Travers. 1976. Mercury in the Atlantic Provinces. EPS, Atlantic Region, Surveillance Report EPS-5-AR-77-10.

34. Baker, D.J. and R.A.F. Matheson. 1980. Cadmium in the Atlantic Provinces. Environment Canada, Environmental Protection Service, Atlantic Region, Surveillance Report EPS-5-AR-80-3.

35. Bryan, G.W. 1976. Some aspects of heavy metal tolerance in aquatic organisms, p. 7-34. *In:* A.P.M. Lockwood (ed.), Effects of Pollutants in Aquatic Organisms, Cambridge University Press.

36. Ray, S., D.W. McLeese and D. Pezzack. 1980. Accumulation of cadmium by *Nereis virens*. Arch. Environm. Contam. Toxicol., 9: 1-8.

37. Ray, S., D.W. McLeese, B.A. Waiwood and D. Pezzack. 1980. The disposition of cadmium and zinc in *Pandalus montagui*. Arch. Environm. Contam. Toxicol., 9: 675-681.

38. Fowler, S.W. and G. Benayoun. 1974. Experimental studies on cadmium flux through marine biota. Radioactivity in the Sea, No. 44, Int. Atomic Energy Agency, Vienna.

39. Ray, S., and D.W. McLeese. Department of Fisheries and Oceans, St. Andrews, N.B. Unpublished data.

40. Schuster, C.N. and B.H. Pringle. 1969. Trace metal accumulation by the American eastern oyster *(Crassostrea virginica)*. Proc. Nat. Shellfish. Assoc., 59: 91-103.

41. George, S.G. and T.L. Coombs. 1977. The effects of chelating agents on the uptake and accumulation of cadmium by *Mytilus edulis*. Mar. Biol., 39: 261-268.

42. Ray, S., D.W. McLeese and D. Pezzack. 1979. Chelation and interelemental effects on the bioaccumulation of heavy metals by marine invertebrates. Proc. Manage. Control. Heavy Met. Env., p. 35-38.

43. Rainbow, P.S., A.G. Scott, E.A. Wiggins and R.W. Jackson. 1980. Effect of chelating agents on the accumulation of cadmium by the barnacle *Semibalanus balanoides* and complexation of soluble Cd, Zn and Cu. Mar. Ecol. Prog. Ser., 2: 143-152.

44. Bryan, G.W. and L.G. Hummerstone. 1973. Adaptation of the polychaete *Nereis diversicolor* to estuarine sediments containing high concentration of zinc and cadmium. J. Mar. Biol. Assoc., U.K., 53: 839-857.

45. Boyden, C. 1977. Effect of size upon metal content of shellfish. J. Mar. Biol. Assoc., U.K., 57: 675-714.

46. Schulz-Baldes, M. 1973. Die Miesmuschel *Mytilus edulis* als indikotor fur die bleikonzentration in Weserastuar und in der Deutschen Bucht. Mar. Biol., 21: 98-102.

47. Ratkowski, D.A., S.J. Thrower, I.J. Eustace and J. Olley. 1974. A numerical study of the concentration of some heavy metals in Tasmanian waters. J. Fish. Res. Board Can., 31: 1165-1171.

48. Bryan, G.W. 1974. Adaptation of an estuarine polychaete to sediments containing high concentrations of heavy metals, p. 123-135. *In:* F.W. Vernberg and W.B. Vernberg (eds.), Pollution and Physiology of Marine Organisms, Academic Press. New York.

49. Bryan, G.W. and H. Uysal. 1978. Heavy metals in the burrowing bivalve *Scrobicularia plana* from Tamar estuary in relation to environmental levels. J. Mar. Biol. Assoc., U.K., 58: 89-108.

50. Goldberg, E.D., V.T. Bowen, J.W. Farrington, G. Harvey, J.H. Martin, P.L. Parker, R.W. Risebrough, W. Robertson, E. Schneider and E. Gamble. 1978. Mussel watch. Environ. Conserv., 5: 101-126.

51. Neff, J.W., R.S. Foster and J.F. Slowey. 1978. Availability of sediment adsorbed heavy metals to benthos with particular emphasis on deposit feeding infauna. Tech. Rep. D-78-42, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss., U.S.A.

52. Ray, S. and D.W. McLeese. 1980. Bioavailability of chlorinated hydrocarbons and heavy metals in sediments to marine invertebrates. Int. Counc. Explor. Sea, C.M.1980/E:20, 13 p.

53. Bryan, G.W. 1981. The biological availability and effects of heavy metals in marine deposits. Presented at the 3rd Int. Ocean Dumping Symp., Woods Hole, Mass.

54. Luoma, S.N. and G.W. Bryan. 1979. Trace metal bioavailability: Modeling chemical and biological interactions of sediment-bound zinc, p. 577-609. *In:* E.A. Jenne (ed.), Chemical Modeling in Aqueous Systems, A.C.S., Symp. Ser. 93, Am. Chem. Soc., Washington, D.C., U.S.A.

55. Langston, W.J. 1984. Comparative studies on the behaviour of As and Hg in United Kingdom estuaries. Can. J. Fish. Aquat. Sci. (in press).

56. Cooke, M., G. Nickless, R.E. Lawn and D.J. Roberts. 1979. Biological availability of sediment bound cadmium to the edible cockle *Cerastoderma edule*. Bull. Environ, Contam. Toxicol., 23: 381-386.

57. Luoma, S.N. and G.W. Bryan. 1978. Factors controlling the availability of sediment bound lead to the estuarine bivalve *Scrobicularia plana*. J. Mar. Biol. Assoc., U.K., 58: 793-802.

58. Pentreath, R.J. 1976. The accumulation of mercury from food by the plaice *Pleuronectes platessa*. J. Exp. Mar. Biol. Ecol., 25: 51-65.

59. Pentreath, R.J. 1977. The accumulation of cadmium by the plaice *Pleuronectes platessa L.* and the thornback ray *Raja clavata* L. J. Exp. Mar. Biol. Ecol., 30: 223-232.

60. Benayoun, G., S.G. Fowler and B. Oregioni. 1974. Flux of cadmium through euphausiids. Mar. Biol., 27: 205-212.

61. Ray, S. and D.W. McLeese. 1982. Notes on bioavailability of cadmium and lead to a marine crustacean through food and water, p. 19-24. Environment Canada, Environmental Protection Service, Surveillance Report EPS-5-AR-82-1.

THE EFFECTS OF HEATED EFFLUENTS ON MARINE WATER QUALITY IN THE ATLANTIC REGION

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Studying the effects of heated effluents on the aquatic environment is a relatively new field of endeavour when compared to the research which has been done with respect to some of the more classical forms of pollution. This is especially true of the marine environment where much less thermal effects work has been done than in fresh water. Because temperature is a non-conservative pollutant, and because marine waters generally have such a vast assimilative capacity, the assumption has been made that the impact of heated effluents on these waters could be of little significance. This is probably a good assumption when considering the open ocean. However, depending on the morphological characteristics of the receiving environment, the volume of heated water discharged and the design of discharge structures, the localized effects of thermal effluents could be significant, even in marine environs.

Temperature affects nearly every physical property of concern in water quality management[1], has a profound effect on chemical reactions in the aquatic environment, and directly affects the physiological processes of most marine organisms (bacteria to fish). Therefore, it is a pollutant which should not be dismissed lightly. In addition, because of man's increasing need for power, there will be a concomitant need for cooling water and hence an increase in the amount of heated effluent released into the marine environment before the end of the century[2]. All of these factors suggest that thermal effluents could be important in determining the quality of some marine waters.

The purpose of this paper is to examine the effects of heated effluents on the marine environment in the Atlantic Region of Canada. This will be done by briefly outlining the types of effects that may be expected in areas influenced by thermal pollution (physical, chemical, biological) and then looking more closely at specific point sources of heated effluent in this Region.

GENERAL EFFECTS OF HEATED EFFLUENTS IN MARINE SYSTEMS

Sources

A large number of man's activities produce waste heat as a by-product. Much of this heat dissipates naturally into the atmosphere, but a significant portion is delivered directly to the aquatic environment as heated effluent. Three major sources of such thermal inputs are: industrial processes (chemical, petro-chemical, pulp and paper); municipal sewage; and thermal generating stations[2]. However, the amount of heated effluent from thermal generation far exceeds the contributions made by other sources of input. It has been estimated, for example, that electricity generation accounts for over 75% of the total heat discharged into the aquatic environment in Canada[2,3]. For this reason, discussions of thermal effects in this paper will relate primarily to heated effluent produced as a by-product of the thermal generation of electricity.

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Areas of Impact

In dealing with heated effluents from thermal generating stations, it should be recognized that there are three potential areas of impact: (1) in-plant effects; (2) near-field effects; and, (3) far-field effects.

In-plant effects are those which result when an organism passes through a thermal power plant with the cooling water. Damage may be imparted to these organisms through mechanical means (impingement on screens, pressure changes) or through rapid increases in temperature as the cooling water passes through the condenser relieving it of its excess heat[5]. It should be noted that further discussion of this type of effect will deal only with those caused by increasing temperature since mechanical damage is not within the scope of these discussions.

Near-field effects are those which occur close to the plant releasing the heated effluent and may be thought of as thermal plume effects. Organisms may be affected by purposely moving into the plume because they are attracted by the warmer water; because the plume may impact an organism that is incapable of avoiding it (e.g., some planktonic and benthic forms) or because a plume may block an organism from behaving normally. Near-field effects are most obvious when the plume exists in a well defined state either because of the design of the outfall structure (e.g., a channel) or because of the dispersion characteristics of the plume in the receiving environment[7].

Far-field effects would not normally be expected in the marine environment. Such effects occur when thermal inputs to a water body gradually raise the temperature of that body to a point where it is above its normal ambient range. Because of the large assimilative capacity of most marine waters in the Atlantic Region, such far-field effects are unlikely to occur[8]. However, it is possible that thermal effluents could be directed into marine bodies which have restricted water exchange rates because of their morphology, etc. In these situations, far-field effects may be expected.

Types of Impact

Temperature affects virtually every physical property of water including density, viscosity, vapour pressure, surface tension and gas solubility and diffusion[1]. In addition, increases in water temperature may also cause an alteration of rate coefficients and equilibrium constants of most chemical processes[6].

Because temperature has such a marked effect on the physical and chemical properties of water, increased temperatures may result in an alteration of the quality of the water in which marine organisms live. These changes may produce a set of conditions which do not favour the community of organisms that normally inhabit that water body. Examples of such alterations might be reduction in the oxygen content of the thermally affected area to a point where organisms are stressed; or, conversely, a chemical pollutant already in the water may become more toxic because of the increased temperature. In either case, the quality of the water could become less suitable for the support of biological communities than it was prior to the introduction of the waste heat.

In addition to the above indirect affects, increased temperatures may also directly affect marine animals. In most cases, aquatic organisms are poikilothermic (i.e., cold blooded) and adjust their own internal physiology to function at the temperature of their environments[6]. Within normal ranges, temperature can have a variety of effects on poikilotherms by acting as a catalyst, depressant, actuator, restrictor, stimulator or controller[1]. The end result of these effects may be displayed in a variety of sublethal responses including changes in metabolic rate, respiration, behavior, growth, reproduction, distribution patterns and susceptibility to parasitism and disease[1]. Within limits, these changes may be beneficial to the affected organism. But if these changes are excessive, negative impacts may result.

When extreme changes occur, the increased temperatures may cause more direct impacts. In cases where they exceed the thermal tolerance of individual species of organism, direct mortality may occur. Such occurrences are rare, however, and would normally happen only in situations where an organism could not avoid the increased temperature and where it was already operating at or near its thermal tolerance limit. Such effects are also determined by the species involved, the thermal history of that particular organism, and the exposure time[9].

An interesting variation on this theme occurs when organisms become acclimated to increased temperatures. This usually occurs only when a thermal plume is confined in some way (e.g., by a release channel). In these cases, animals can become dependent on the warmer water of the thermal plume and if for some reason the source of heat is removed (e.g., shutdown of plant for maintenance), the rapid return to normal ambient temperatures may be enough to induce thermal shock. Fish kills near thermal plants are normally related to this phenomenon[10,11].

Finally, thermal plumes may act as barriers to the natural migrations of marine organisms between nearshore and offshore waters, to diurnal vertical migration for feeding, or to seasonal migration for spawning[8].

Regardless of whether marine organisms are affected by external physical or chemical changes, internal physiological changes (lethal or sublethal) or by being hindered from some activity because of an avoidance reaction, the overall effect on these exposed organisms may be such that there is a shift in population structure of the impacted communities[12]. Such shifts may occur if a significant portion of a particular planktonic community is passed through a cooling water system, may be the end result of a benthic community being continually exposed to a thermal plume of elevated temperatures, or may be due to the effects of a confined plume on organisms that acclimate to the warmer water. In any case, an overall reduction in community diversity is possible, concurrently resulting in a reduction in the "health" of that particular community[13].

The environmental effects outlined above are very general in nature and could apply to any body of water (fresh or marine) receiving thermal inputs. In order to determine actual effects related to a specific location, it is necessary to study the details of that particular site. Information would be necessary on the amount of heat input, the temperature change of the cooling water, the morphometry of the receiving water body, the cooling water system, and the communities of organisms exposed to the changing thermal regime. Only with such information in hand can the impact of a specific heat source be determined.

HEATED EFFLUENTS IN THE ATLANTIC REGION

This section will generally describe the sources of heated effluents in Atlantic Canada. As was the case in Section A, this discussion will be restricted to thermal generating stations since these sites provide by far the greatest source of waste heat released into marine waters[2,3].

The total number of thermal generating stations in Atlantic Canada which release thermal effluents to the marine environment is thirteen. The distribution of these plants is shown in Figure. 1. As noted previously, there has not been a great deal of concern with respect to heated effluents released into the marine environment and, as a consequence, the information available describing thermal effects in the Region is rather sparse.*

Nova Scotia

There are six thermal generating stations in Nova Scotia which discharge heated effluent to the marine environment. These plants range in size from 165-350 Megawatts (MW) installed capacity and generally raise the temperature of their cooling water between 6.1 and 14.4°C. Half of these plants discharge heated effluent through surface channels whereas the other 50% use submerged pipelines to carry the heated water offshore. Environmental monitoring has been conducted only at the Glace Bay station in Cape Breton, and this only to delineate the thermal plume.

No survey or monitoring for environmental effects have been undertaken in relation to this plant.

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* The information outlined in this section was taken primarily from a review of a "Steam-Electric Generating Station Inventory" which was prepared by Environment Canada in 1976.

New Brunswick

There are 4 thermal generating stations in New Brunswick which release heated effluent to the marine environment, including the nuclear generating station at Point Lepreau. The installed capacities of these plants range from 322 to 1050 MW and the temperature of the cooling water is raised an average of 10.7°C. All of these plants utilize outfall pipes which discharge the heated effluent at submerged discharge points offshore.

Some environmental monitoring has been conducted with respect to the Dalhousie thermal generating station and an extensive monitoring program is presently underway at Point Lepreau.

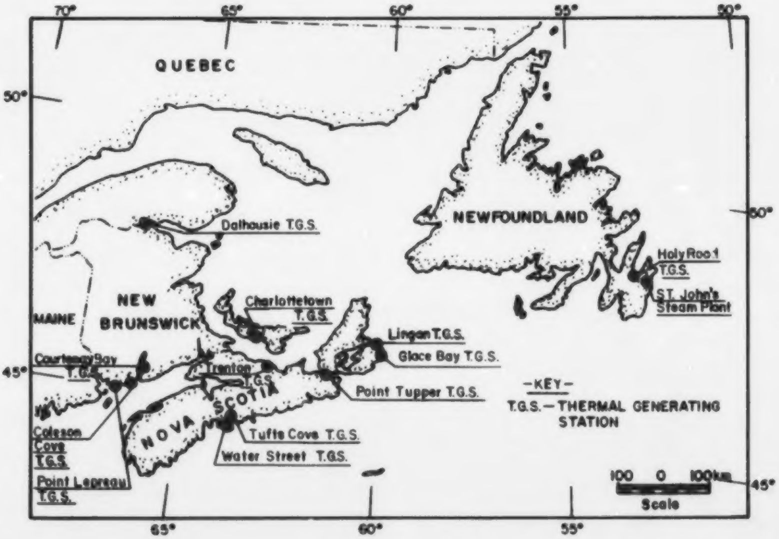
Newfoundland

Newfoundland has two thermal generating stations, one in St. John's and the other in Holyrood with installed capacities of 30 and 450 MW, respectively.

Some environmental monitoring has been done at the Holyrood site[16].

Prince Edward Island

The only thermal generating station in Prince Edward Island is the Charlottetown generating station situated on Charlottetown Harbour. This


70°
65°
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60°
55°
QUEBEC
NEWFOUNDLAND
Dalhousie T.G.S.
HolyRood T.G.S.
ST. John's Steam Plant
NEW BRUNSWICK
MAINE
Charlottetown T.G.S.
Lingan T.G.S.
Glace Bay T.G.S.
Courtenay Bay T.G.S.
Trenton T.G.S.
Point Tupper T.G.S.
-KEY-
T.G.S.—THERMAL GENERATING STATION
NOVA SCOTIA
Coleson Cove T.G.S.
Tufts Cove T.G.S.
Water Street T.G.S.
Point Lepreau T.G.S.
100 0 100km
Scale

Figure 1: *Distribution of thermal generating stations (T.G.S.) in the Atlantic Region.*

plant has an installed capacity of 70 MW (7 units) and discharges its heated effluent directly into Charlottetown Harbour. No further data on the quantity or quality of the heated effluent is available. Nor is there any information available on the environmental effects of this discharge.

SUMMARY AND CONCLUSIONS

The foregoing review would seem to indicate that thermal pollution in the Atlantic Region is not a significant factor in the determination of marine water quality. Although very little empirical data on environmental effects is available in relation to thermal generating stations in this region, the limited information on physical dispersion of heated water would suggest that the potential areas of impact are quite limited (1-25 ha). It would also seem that even though very few formal studies have been undertaken, those that have been done indicate that there is no evidence for the existence of major environmental impact.

The lack of information concerning environmental quality should not be construed to mean that no effects are present. Rather, this could indicate the need for comprehensive studies of environmental impacts at some of these locations if a detailed picture of environmental quality is needed. Perhaps the studies presently underway in relation to the Pt. Lepreau nuclear generating station will more clearly define the need for additional studies with respect to other generating stations.

One thing is clear. Over the next two decades, the need for power is going to increase[2], and with this increased need there is going to be a concomitant increase in the amount of waste heat produced by generating stations. Whether this increased effluent is produced by existing facilities, or by new facilities, there is an obvious need to more clearly define the potential problems related to the heat production. Marine water quality can only be maintained (or improved) if there is knowledge of the effects of heated effluents and sound planning with respect to the deposition of these effluents into the marine environment.

REFERENCES

1. Parker, F.L. and P.A. Krenkel. Thermal pollution: status of the art. National Centre for Research and Training in the Hydrologic and Hydraulic Aspects of Water Pollution Control. Report No. 3 (1969).

2. Montreal Engineering Co. Ltd. The thermal inputs into Canadian waters excluding the Great Lakes Basin 1970-2000 A.D. Report produced under contract to Environment Canada (1971).

3. Denison, P.J. and Elder, P.J. Thermal inputs to the Great Lakes 1968-2000. Proc. 13th Conf. Great Lakes Res. pp 811-828 (1970).

4. James F. MacLaren Ltd. Review of heated discharge management and control alternatives (once-through systems in large water bodies). Environment Canada Report Series, Report No. EPS-3-WP-75-4 (1975).

5. Schubel, J.R. and B.C. Marcey, Jr. (Eds.). Thermal effects of entrainment *In:* Power plant entrainment, a Biological Assessment. Academic Press, New York (1978).

6. Dickson, D.R. Waste heat in the aquatic environment. National Research Council, Associate Committee on Scientific Criteria for Environmental Quality.

7. Thorburg, A. Biological effects of thermal effluents in the marine environment: tropics and sub-tropics with a guideline. FAO Report, Club of Rome (1980).

8. Environment Canada. Environmental codes of practice design phase manual, Part D, Rationale for recommendations, once-through cooling systems (Unpublished).

9. U.S. National Academy of Science and National Academy of Engineers. Water quality criteria, 1972. Environmental Studies Board, Washington, D.C. EPA-R3-73-033 (1973).

10. Ash, G.R., N.R. Chymko, and D.N. Gallup. Fish kill due to cold shock in Lake Wabamun, Alberta. J. Fish. Res. Bd. Canada 31: 1822, (1974).

11. Cairns, J., Jr. Coping with heated waste water discharges from steam-electric power plants. Bioscience 22: 411-420 (1972).

12. Cole, R.A. Stream community response to nutrient enrichment. J. Water Poll. Control Fed. 45(9) 1874 (1973).

13. Tenore, K.R. Macrobenthos of the Pambico River Estuary, North Carolina. Ecological Monographs. 42, 51 (1970).

14. MacLaren Atlantic Ltd. Marine bottom communities in the vicinity of the Dalhousie generating station discharge. Unpublished Report to Environment Canada.

15. MacLaren Atlantic Ltd. Environmental assessment for the Point Lepreau generating station. Unpublished Report to the New Brunswick Electric Power Commission (1977).

16. Whittick, A. and R.G. Hooper. A limited study on the effects of the thermal effluent from the Holyrood generating station on the plant and animal benthos of Conception Bay. Unpublished Report for the Centre for Research in Labrador, Memorial University of Newfoundland.

17. Leblanc, P. Nova Scotia Power Corporation, Personal Communication.

PERSISTENT LITTER

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INTRODUCTION

Apart from oil pollution, persistent litter is one of the only kinds of marine pollution that is readily visible in the marine and coastal environments. Other forms of pollution such as chemicals, heat, sewage, and suspended material usually require sensitive instruments to detect their presence, since they manifest few symptoms to the casual observer. Because of this, the "health" of the oceans, as perceived by the layman, often varies according to the amount of floating or stranded litter seen along the coast. Despite the high visibility of litter in the sea, its importance in relation to other contaminants is low and it has, for the most part, been quite neglected from a scientific point of view.

This paper will discuss the kinds of materials that make up persistent litter and the sources of these materials, as well as some of the environmental problems that they cause. Very little data is available for marine waters of the Atlantic Region, thus inferences must be drawn from studies elsewhere. Information is based mainly on a handful of studies and a number of reported observations from various locations around the world. Although pelagic tar and tar balls are ubiquitous and have been treated by some as litter, they will not be discussed in this section.

For the purpose of this paper, persistent litter can be defined as any material that is lost, discarded, dumped or discharged into the marine environment, or that blows into the sea, or is carried down rivers and ends up in the sea. The only other criterion is that the material must be resistant to rapid degradation or breakdown in the environment. Materials such as paper, hemp rope, small vegetation, wood chips, etc., are not persistent, whereas plastics, metals and glass are broken down only very slowly in the marine environment and are thus considered persistent.

KINDS OF MATERIALS

Persistent litter contaminates the world's oceans in two major ways: by floating at or near the surface due to a neutral or slightly positive buoyancy, or by sinking and cluttering the sea floor. Another problem associated with persistent litter is the fouling of shoreline and beaches, which is often the final fate of floating debris.

Plastic litter in the marine environment has, during the last decade, been recognized as a concern mainly because of the widespread use of plastics and their intrinsic properties. Plastics usually have a low specific gravity, between 0.8 and 0.96, and thus float. They are, for the most part, non-biodegradable and only slowly photodegrade when exposed to ultraviolet radiation. Also, some plastics contain additives which stabilize them

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against ultra-violet light and act as anti-oxidants, thus greatly extending their existence in the environment.

Various kinds of plastic have been found at sea in a wide variety of forms, including both raw materials and fabricated products such as toys, ropes, nets. etc. Raw plastic from plastic producers is shipped to processors in the form of pellets or "nibs", which are usually spheres, discs or cylinders, and which measure between 1 and 5 mm in diameter. Plastic pellets of this type have been found in the coastal waters of New England[1], in the Mediterranean[2], in the Gulf of Alaska and Bering Sea[3], in surface waters of the South Atlantic[4], and Northwest Atlantic oceans, throughout the Caribbean[5] and in the Sargasso Sea[6]. They have also been found on beaches in parts of the South Pacific Ocean[7] and the eastern seaboard of the United States[8]. These plastic pellets are found to consist of one or other of the following: polystyrene, polyethylene, polymethyl methacrylate, and polypropylene, and because of their specific gravity or gaseous inclusions they usually float.

Fabricated plastic products contribute to a more widespread and visually offensive contamination of the open sea and shoreline. The rapidly expanding use of plastics during the 1960s and 1970s, particularly for packaging, containers and disposable items, has led to a rapid proliferation of litter in the world's oceans[9]. On two isolated stretches of shoreline adjacent to the North Sea, Scott[9] collected items, mainly from Europe and North America, which fall into three main categories: domestic and industrial waste, agricultural packaging, and maritime waste. Similar beach surveys carried out on the English Channel[10], and on a Florida beach[11], showed much the same results, with blow-moulded plastic containers being the most abundant item. A survey of the shoreline in Narraganset Bay, on the other hand, indicated that most of the litter originated from the recreational use of the shore of coastal waters, since items such as styrofoam cups, milk shake tops, beer can carriers, plastic bags, shotgun pellet holders, and paper and cardboard products were prevalent in the litter[12]. Floating litter encountered at sea included fragments of styrofoam and pieces of hard and soft plastic from containers, toys, etc., as well as polyethylene sheeting[5]. Also encountered are fishing floats, rope, fragments of fishing nets, wood, and metal and glass containers[13].

Quantities of plastic fragments vary from one per 9,000 m² in the Gulf of Alaska and Bering Sea[3], to one piece per 280 m² in the Sargasso Sea[6], and one piece per 29 m² in the vicinity of 143° W; 35° N in the Pacific Ocean. Obviously, the distribution of such objects is non-uniform and concentrations vary according to the amount of shipping traffic, proximity to populated coastal areas, and the structure of oceanic currents, location of confluences, and characteristics of surface winds[14,15].

Much of the debris dumped, discarded or lost at sea does not remain at the surface, but sinks to the bottom or drifts at some intermediate depth, depending on its density. Such things as metal, glass and plastic containers, rope and twine, wire, metal fragments, cloth, rubber, fishing gear, wood and paper products have been collected in trawls in the Bering Sea[16] and the Gulf of Alaska[17]. In these studies, debris of this kind was collected in 40-60% of the trawls, with the most common types being plastic, metal, rope and twine. Large quantities of fishing gear (ropes, nets and floats) have been noted on beaches and shorelines of the Aleutian Islands in the North Pacific Ocean[18], and it follows that lost or discarded fishing gear probably makes up a large proportion of the bottom debris as well.

A definite gap exists in quantitative knowledge of debris on the sea floor. The ubiquitous plastic pellets, reported to be so prevalent on beaches around the world, also apparently sink (some have a density greater than 1.0) and have been found in sediment samples at levels as high as 20,000 pellets per m^2 close to industrialized centers[19]. Accounts of larger objects or structures accidentally lost or discarded at sea include drill pipe and drill heads, heavy wires, and hawsers[20]. The environmental implications and human risk from such objects will be discussed in a later section.

SOURCES OF LITTER

The sources of persistent litter in the oceans are extremely diverse, and arise from a number of factors, including: increased use of plastics in wrappings, containers, and disposable items; development of fishing nets and ropage made from plastic products such as polypropylene; careless discharge of garbage from ships at sea and from urban centers along the coast; and increased use of marine waters for recreation.

A high proportion of marine litter is discarded at sea as shipboard-generated solid wastes, which are produced in quantities ranging from 1.1 to 2.6 kg per person per day, with an additional 290 tonnes of cargo-associated wastes per ship per annum. Assuming that solid wastes from all ships are disposed of overboard, then this represents an estimated 6.5×10^6 tonnes of litter per annum, mostly distributed within 400 km of land in the northern hemisphere[21]. Although only 0.7% of this litter is plastic, it floats, is extremely persistent, and creates the most serious aesthetic and environmental problems[18].

The fishing industry is probably one of the major offenders. With the introduction of plastics into fishing gear in the past couple of decades, virtually all ropes and nets are now made of polypropylene and other non-degradable polymers. In the North Pacific, 98% of the total weight of debris found on beaches was commercial fishing gear. Trawl web was the most common, followed by polypropylene rope, trawl floats, gillnet floats, and polyethylene bulk containers. The annual rate of increase in beached litter in the North Pacific was 50% between 1972-1974, 10 times more than hypothetical estimates. This huge increase is attributable to the operation of large fishing fleets and attendant accidental loss and deliberate dumping of discarded plastic fishing gear[18]. Between 1962 and 1974, an annual average of 697 Japanese vessels and 453 Soviet vessels fished in this area off the coast of Alaska. On the Atlantic coast in or near Canadian waters (200 mile limit), there are approximately 344 foreign vessels licenced or operating. The licenced domestic fleet numbers 32,000 vessels, but only about 400 are over 65 feet in length.

There is considerable evidence that debris lost or discarded by the oil industry in exploration, drilling and well production has created problems for shipping and fishing, particularly in the North Sea where these activities are concentrated. Material such as drill heads, pieces of steel piping, wire and hawsers of various description, empty drums and paint cans have been found in the vicinity of oil and gas installations[20]. These and other types of debris have also been attributed to the activities of offshore supply vessels.

One of the common items littering the Arctic waters of the Beaufort Sea, where extensive seismic work is underway, is the plastic explosive charge container. These items float and are eventually stranded on beaches where they accumulate in large numbers[22]. At the present time, there is no major source on the Atlantic coast, since the majority of seismic work is done using the "air-gun" technique.

The discharge of municipal and industrial wastes into river estuaries and the marine environment inevitably contributes to coastal litter. For example, one item of prominence littering many beaches in the Atlantic Region is the plastic tampon dispenser, evidently discarded into the sewer system and discharged to marine waters along with untreated sewage. The prevalence of raw plastic pellets near industrial centers such as the Bristol Channel[19], and adjacent to plastic manufacturers[6,23] also indicates careless or accidental discharge of this raw material to the marine environment along with other industrial waste. Elastic threads found in the digestive tracts of puffins in the English Channel appear to be discarded waste of the garment industry[24], either dumped at sea or discharged in effluent from this industry.

Due to the northward movement of ocean currents such as the Guyana and Antilles currents in the vicinity of northern South America, and the Caribbean Current, quantities of debris (mostly plastic litter) are dispersed into the southern and mid-North Atlantic along the coast of North America. Some is picked up by the Gulf Stream and moved even further north. A beach survey, conducted at Fort Pierce in Florida, identified plastic items from the United States as well as Venezuela, Colombia, Guatemala and Jamaica[11].

Litter from recreational boating and general use of the sea shore can be significant in areas of heavy use such as beaches, marinas, campgrounds, etc., near highly populated areas. For the most part, this litter is localized and usually more of an aesthetic or nuisance problem rather than a serious environmental hazard. However, in view of the fact that plastics production is doubling every 12 years, and society is becoming more and more committed to throw-away or disposable common every-day items, the problem of coastal litter from recreation will continue to grow.

ENVIRONMENTAL PROBLEMS

There are a number of environmental problems that can be attributed to persistent litter in the oceans; however, it is generally accepted that, environmentally speaking, the problems are minor in comparison with toxic substances, petroleum products, sewage waste and other such contaminants. There are, however, a number of human safety factors, environmental hazards, and aesthetic problems associated with litter, which will be discussed here.

Aesthetic Problems

The unaesthetic and annoying features of marine litter are familiar to anyone who has walked a beach anywhere in the world. The ubiquity of beach litter is astounding. The quantity of this type of litter is also shocking; for example, 536 items of plastic, with a total weight of 345 kg, were collected per km of beach on Amchitka Island in the North Pacific in 1974. This represents the accumulation of persistent litter for only one year, even in this relatively remote part of the North Pacific Ocean[18].

Apart from decreasing the aesthetic and recreational enjoyment of the shoreline, some litter such as pop-tops, broken glass and rusty nails can cause cuts or injuries and thus are hazardous to human visitors to beaches. Similar aesthetic problems result from debris floating in open ocean water. Venrick[13] calculated, from observations made in the central North Pacific Ocean in 1973, that over 5 million plastic bottles were afloat in the Pacific Ocean at that time. Considering their long life in the environment, and the annual production of such bottles (5×10^9), this number is conceivable.

Hazard to Marine Life

Entanglement seems to be one of the most serious environmental problems arising from litter in the oceans, and marine mammals appear to be the most susceptible victims. Floating marine litter apparently attracts marine mammals, and teeth marks in plastic bottles and other containers probably come from testing for food or in play. Similar attraction to nets or plastic loops or straps can result in seals becoming entangled. Seals trailing netting, fishing line, or girdled by plastic straps have become common occurrences in the Pribilof area of the North Pacific. Entangled northern fur seals have increased from 0.17% of the harvested males in 1967 to 1.55% in 1977. Over 30% of these entanglements are attributed to the circular plastic straps used on bundles of new netting, which when discarded uncut, can become lodged around the head, neck or shoulders of seals. The data above are based on seals that survive to return to the breeding area. Unknown numbers are fatally entangled and die of drowning, malnutrition, or from wounds inflicted by the entangling material cutting into their flesh. In the North Pacific Japanese gillnet salmon fishery alone, more than 3,000 fur seals are killed annually by entanglement, indicating the potential hazard of derelict or discarded gillnets to these mammals[18]. Similar problems are described for the Cape fur seal in the South Atlantic, where about 0.1% of the annual harvest were found to be entangled in fishing net, rope, rubber o-rings, wire, monofilament and plastic straps[25]. Much less common, but reported, are fiberglass straps on sharks, net fragments on baleen whales, and rope collars on Antarctic fur seals. It would appear, from discussion with fisheries personnel and research scientists, that the problem of sea mammal entanglement with plastic litter along the Atlantic coast is uncommon. On Sable Island, where observations of grey and harbour seals have been carried out for many years, only 2 or 3 incidents are noted per year out of a yearly count of some 12,000 grey seals and about 500 to 700 harbour seals combined. Most incidents involve fragments of trawl net. Harp seals are rarely, if ever, found entangled, possibly due to their high northern distribution near ice in areas less frequented by the fishermen.

The ingestion of plastic pellets and rubber fragments by marine fish and birds has been documented; however, the hazard to these organisms has not been well defined. Polystyrene spherules have been found in the stomachs of a number of species of larval and juvenile fish. In Niantic Bay, off New England, eight species of fish and one chaetognath were found to contain plastic spherules in their guts[1]. It is speculated that this material could cause intestinal blockage in some smaller fish; for example, winter flounder larvae 5 mm in length had spherules 0.5 mm in diameter in their guts. If mortality and loss results from such occurrences, then the observed numbers so affected might be much fewer than the actual number. Larval and juvenile fish surveyed by Colton *et al.*[5] were found to contain no plastic particles, so the problem is not clear and requires more investigation.

Plastic particles and elastic threads have been found in tern pellets (droppings) and the digestive tract of puffins, respectively. The polystyrene spherules ingested by terns were probably first swallowed by small fish, which were then eaten by the terns. There is no evidence that these particles do any damage to the terns[8]. In the case of rubber threads eaten by puffins, in some instances the entangled elastic was tightly packed into the gizzard exit or formed a ball of rubber in the gizzard itself. Either of these situations could lead to blockage of the digestive system, or at least reduced efficiency of the gizzard in grinding up food[24]. Indication that there might be a detrimental effect of plastic pellets in the stomachs of birds has been documented by Connors and Smith[26], who examined seven red phalarope *(Phalaropus fulicarius)* from a flock of 6000 migrants on the coast of California. In some birds as much as 20% of the stomach volume was taken up by plastic pellets. A negative correlation between amount of plastic and fat condition of the birds suggested a detrimental effect. An item which has recently been introduced into marine litter is the plastic "six-pack" beer can retainer. With the cans removed the holes in the retainer have been reported to entrap the necks, wings or legs of gulls, resulting in immobilization and death.

Potential toxicity from plastic litter in the oceans is a matter of speculation. Numerous additives are included in plastics to control their characteristics: plasticizers to improve flexibility, colour pigments, antioxidants, ultra-violet light stabilizers, lubricants, mould-release agents and biocides to inhibit microbial degradation, slip agents and anti-block agents to prevent sheets from sticking together, and antistatic agents[27]. The release of these additives to the environment from plastic litter has not been adequately studied. Some plastic particles have been analyzed for contaminants such as PCB, and levels up to 5 ppm have been detected in spherules of polystyrene[1]; however, in this case, the PCB was apparently absorbed from the environment and not added during manufacture. PCB might be added to plastics during production of plastic items; however, the potential impact on levels of this and other contaminants in the environment from discarded plastic is unknown.

Hazard to Ships and Fishermen

Floating or sunken debris can present a serious or nuisance hazard to ships and mariners. Debris can put fishing vessels and crew at risk either by tangling gear on sea bed obstructions, or by fouling propellers. In both cases, the vessel would not be fully manoeuvreable, an extremely dangerous situation in bad weather. In the former situation there is the additional danger of the vessel capsizing if action is not taken promptly to free the gear by cutting it loose. Also, if the gear breaks under severe strain, injuries to deck crew could result from the whiplash effect of the wires being released from the strain[20].

Heavy objects, such as logs or wood structures floating at or near the surface, pose a real hazard to small boaters and can also seriously damage larger vessels passing over them. Sheets of plastic have been known to block cooling water intakes for ships' engines, resulting in damage and loss of operating time. Hazard to ships and crew in the waters around the U.K. has reached the point where large sums of money are paid in compensation for lost gear and "hardship" to vessel owners and crews. Both in the North Sea and in Canadian Waters, authorities are trying to institute compulsory indelible marking of the oil companies' gear (rope, plastic items, cables, etc.) so that incidents of ocean littering can be traced to the offenders.

REFERENCES

1. Carpenter, E.J., S.J. Anderson, G.R. Harvey, H.P. Miklas and B.B. Peck. 1972. Polystyrene spherules in coastal waters. Science 178: 749-750.

2. Shiber, J.G. 1979. Plastic pellets on the coast of Lebanon. Mar. Pollut. Bull. 10: 28-30.

3. Shaw, D.G. 1977. Pelagic tar and plastic in the Gulf of Alaska and Bering Sea: 1975. Sci. Total Env. 8: 13-20.

4. Morris, R.J. 1980. Plastic debris in the surface waters of the South Atlantic. Mar. Pollut. Bull. 11: 164-166.

5. Colton, J.B., F.D. Knapp and B.R. Burns. 1974. Plastic particles in surface waters of the northwestern Atlantic. Science 185: 491-497.

6. Carpenter, E.J. and K.L. Smith. 1972. Plastics on the Sargasso Sea surface. Science 175: 1240-1241.

7. Gregory, M.R. 1977. Plastic pellets on New Zealand beaches. Mar. Pollut. Bull. 8: 82-84.

8. Hays, H. and G. Cormons. 1974. Plastic particles found in tern pellets, on coastal beaches and at factory sites. Mar. Pollut. Bull. 5: 44-46.

9. Scott, G. 1972. Plastic packaging and coastal pollution. Internat. J. Environ. Studies 3: 35-36.

10. Dixon, T.R. and A.J. Cooke. 1977. Discarded containers on a Kent beach. Mar. Pollut. Bull. 8: 105-108.

11. Winston, J.E. 1982. Drift plastic - an expanding niche for a marine invertebrate? Mar. Pollut. Bull. 13: 348-351.

12. Cundell, A.M. 1973. Plastic materials accumulating in Narraganset Bay. Mar. Pollut. Bull. 4(12): 187-188.

13. Venrick, E.L., T.W. Backman, W.C. Bartram, C.J. Platt, M.S. Thornhill and R.E. Yeats. 1973. Man-made objects on the surface of the central North Pacific Ocean. Nature 241: 271-272.

14. Shaw, D.G. and G.A. Mapes. 1979. Surface circulation and the distribution of pelagic tar and plastic. Mar. Pollut. Bull. 10: 160-162.

15. Wong, C.S., D.R. Green and W.J. Cretney. 1974. Quantitative tar and plastic waste distributions in the Pacific Ocean. Nature 247: 30-32.

16. Feder, H.M., S.C. Jewett and J.R. Hilsinger. 1978. Man-made debris on the Bering Sea floor. Mar. Pollut. Bull. 9: 52-53.

17. Jewett, S.C. 1976. Pollutants of the northeast Gulf of Alaska. Mar. Pollut. Bull. 7: 169.

18. Merrell, T.R. 1980. Accumulation of plastic litter on beaches of Amchitka Island, Alaska. Mar. Env. Res. 3: 171-184.

19. Morris, A.W. and E.I. Hamilton. 1974. Polystyrene spherules in the Bristol Channel. Mar. Pollut. Bull. 5: 26-27.

20. McCall, R.A. 1980. Consequences for fishing industry of oil industry debris. Marine Policy, January: 61-63.

21. Dixon, T.R. and T.J. Dixon. 1981. Marine litter surveillance. Mar. Pollut. Bull. 12: 289-295.

22. Wong, C.S., D. MacDonald and W.J. Cretney. 1976. Tar and particulate pollutants on the Beaufort Sea coast. Beaufort Sea Tech. Rept. No. 13. Beaufort Sea Project, Victoria. 96 pp.

23. Gregory, M.R. 1978. Accumulation and distribution of virgin plastic granules on New Zealand beaches. N.Z. J. Mar. Freshwater Res. 12: 399-414.

24. Parslow, J.L.F. and D.J. Jefferies. 1972. Elastic thread pollution of puffins. Mar. Pollut. Bull. 3: 43-45.

25. Shaughnessy, P.D. 1980. Entanglement of Cape fur seals with man-made objects. Mar. Pollut. Bull. 11: 332-336.

26. Connors, P.G. and G. Smith. 1982. Oceanic plastic particle pollution: Suspected effect on fat deposition in red phalaropes. Mar. Pollut. Bull. 13: 18-20.

27. Cundell, A.M. 1974. Plastics in the marine environment. Environmental Conservation 1: 63-68.

MICRO-ORGANISMS AND PATHOGENS

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MICRO-ORGANISMS AND PATHOGENS

Micro-organisms play an important role in the transformation and mineralization of organic substances in a marine environment. By the breakdown of organic pollutants, micro-organisms contribute decisively to the natural self-purification of the sea but, on the other hand, some micro-organisms may produce undesirable effects by causing epidemics through infections.

The Role of Micro-Organisms in the Self-Purification of Waters

Most coastal waters of Atlantic Canada are almost continuously exposed to pollution by either domestic, agricultural or industrial discharges. Hence, the natural self-purification of marine waters is extremely important. This process keeps removing pollutants from the waters so that coastal areas are safe for recreational and fishery use. Physical and chemical processes like sedimentation and oxidation play an important part, but the decisive role must be attributed to biological processes. Micro-organisms decompose organic compounds into carbon dioxide, water and inorganic salts. Protein and carbohydrate are broken down particularly quickly; fat, wax, cellulose and lignin much more slowly, and sometimes incompletely. The rate of purification in an ecosystem is dependent on the microflora, physical factors and the time of year. Microbial degradation is much greater during the summer months than in winter because the metabolic activity of most micro-organisms is greater during the warm season.

The impact of the action of contaminants on certain microbial activities may vary according to the environment. There are differences in the pattern of degradation among contaminants. Most substances have either an inhibitory or a stimulatory effect, depending on their concentration and type. Most domestic sewage, pesticides and oils will degrade with time. Other contaminants, such as heavy metals, are non-degradable, although solubilization and volatilization may eventually reduce amounts in a given environment. However, biodegradation functions only under conditions where composition and quantity of contaminants do not overtax the power of self-purification of the receiving body of water. Excessive discharge of organic wastes in a coastal zone may result in the depletion of oxygen content, and thus create an anaerobic zone, where putrefying processes and bacterial sulphate reduction lead to the production of hydrogen sulphide. This, in turn, causes the death not only of almost all higher organisms, but also of many micro-organisms. Consequently, a microbial population

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consisting of only selective species develops, which can only partially break down the organic pollutants.

Micro-organisms play an important part in the degradation of oil spills. Hydrocarbon-decomposing bacteria and fungi degrade water soluble components of the oil very quickly. Their activity depends on water temperature and concentration of inorganic nutrients, particularly phosphorous and nitrogen compounds[1]. Flat slicks of oil, called lawns, are broken down very slowly by microbes because they can only be attacked from the surface. Fortunately, crude oil alters rapidly in the sea, mainly due to physical processes. Within 14 days, the lighter components volatilize, so that almost the only components which remain are those which are not easily volatilized. These mix with the seawater to form a brown, mushy suspension whose greater surface area improves the chances for bacterial attack. For that reason, artificial emulsifiers are frequently used in combating oil pollution. But most emulsifiers are usually toxic to many aquatic organisms and, therefore, may cause more damage than the oil itself. Some of these emulsifiers inhibit bacterial growth and, as a result, instead of aiding the microbial decomposition of oil, they inhibit it[2].

The microbial degradation process may be disturbed by the introduction of toxic substances from industrial effluents. Some industrial effluents may contain toxic chemical compounds that are inhibitory to microbial metabolic activity, or may even cause death to the micro-organisms involved in the remineralization processes. This happens particularly with compounds of heavy metals, cyanides and chlorinated hydrocarbons[3,4].

PATHOGENS IN WATERS

The discharge of sewage into coastal waters has several potentially adverse consequences, one of which is the causation of human infections. Numerous pathogenic and non-pathogenic intestinal micro-organisms are contributed by humans and other warm-blooded animals to sewage which enters the marine environment through untreated or incompletely treated municipal and hospital waste, private disposal systems, and agricultural or stormwater runoff. Pathogenic micro-organisms can survive and may remain virulent for a period of time in the marine environment. Hence, they can still cause illness, usually through contact with the water or ingestion of contaminated shellfish. The most common waterborne pathogens include strains of *Salmonella, Shigella, Pseudomonas, Leptospira,* enteropathogenic *E. coli, Pasteurella, Vibrio, Mycobacterium,* enteric viruses, parasites *(Entamoeba histolytica* and hookworm larvae), and fungi *(Candida albicans).* The incidence of pathogens in waters is dependent on the source of pollution, the degree of sewage treatment provided, and the level of infectious diseases of the tributary population. Table 1 lists the most important pathogens in waters and their associated diseases. A brief discussion of the significance of the pathogens listed in Table 1 follows.

TABLE 1 - MAJOR WATERBORNE PATHOGENS AND THEIR ASSOCIATED DISEASE

Group	Associated Disease
1. *Bacterial Agents*	
Salmonella sp.	Typhoid, Paratyphoid and Salmonellosis
Shigella sp.	Dysentery
Vibrio sp	Cholera, Gastroenteritis
Leptospira sp.	Leptospirosis
Pasteurella sp.	Tularemia
Mycobacterium sp.	Tuberculosis
Enteropathogenic E. coli	Gastroenteritis
2. *Enteric Viruses*	
Hepatitis	Infectious hepatitis
Poliovirus	Paralytic poliomyelitis and Aseptic meningitis
Coxsackie Virus A	Herpangina and Meningitis
Coxsackie Virus B	Pleurodynia, Meningitis and Infantile myocarditis
ECHO Virus	Aseptic meningitis and Gastroenteritis
Adenovirus	Respiratory and eye infections
Reovirus	Respiratory infection and Diarrhea
Viral Gastroenteritis	Gastroenteritis
3. *Parasitic Agents*	
Entamoeba histolytica	Amebiasis
Giardia lamblia	Diarrhea
Naegleria gruberi	Amoebic meningocephalitis
Ascaris lumbricoides	Ascariasis (hookworm)
Taenia saginata	Taeniasis (tapeworm)
Schistosoma sp.	Schistosomiasis (liver fluke)
Ancylostoma americanus	Ancylostomiasis (hookworm)
Diphyllobothrium latum	Diphyllobothriasis (tapeworm)
4. *Fungi*	
Candida albicans	Oral and Vaginal infections
Trichophyton sp.	Skin infection

Salmonellosis

Salmonellosis is the most frequent water-borne bacterial infection. Samonellae are frequently detected in the marine environment as a result of contamination by human and animal feces. At least several hundred *Salmonella* serotypes are known to be pathogenic to man. The etiologic agent for typhoid fever is *Salmonella typhi*. Typhoid is specific for man, and it does not occur in animals like most other *Salmonella* strains. *Salmonella* infections are contracted by drinking or swimming in sewage contaminated waters, or consuming raw molluscan shellfish harvested from polluted shellfish growing waters.

Salmonellosis is a disease ranging in severity from mild gastroenteritis to septicemia, enteric fever and meningitis. According to Woollen[5], salmonellosis is an increasing world problem, despite efforts at surveillance, whose incidence is grossly under-reported throughout the world. In England and Wales, the number of human salmonellosis cases doubled between 1966 and 1971. In the United States, about two million human salmonellosis cases, 500 deaths and related economic losses of 100 million dollars annually, are attributed to *Salmonella* infections. In Canada, it is estimated that about 5,000 cases are diagnosed annually. However, it is suspected that the reported cases most probably represent only about one percent of the total *Salmonella* infections[6].

Shigellosis

Shigella is the causative agent for acute diarrhea (dysentery). Symptoms of shigellosis may vary from a mild transitory diarrhea to severe prostrating attacks, accompanied by high temperatures, vomiting and profuse bloody stools. Most shigellosis epidemics are food-borne or spread by person-to-person contact. However, there have been a significant number of epidemics which resulted from drinking contaminated waters. These outbreaks frequently result from accidental breaks in water treatment systems, cross connection of contaminated water pipes, or through sewage line seepage into water supply lines. In August, 1974, 31 cases of shigellosis were reported in Dubuque, Iowa, from swimming in the Mississippi River[7]. Experimental data showed that a small number of ingested Shigellae (10 to 100 organisms) is sufficient to produce disease[8].

Cholera

The bacterial pathogen *Vibrio cholerae* can produce a serious acute intestinal disease that is characterized by sudden diarrhea, vomiting, dehydration, temperature and complete collapse. Death may occur within a few hours of onset if prompt medical treatment is not received. Cholera may be spread through person to person contact or consumption of contaminated food and waters. Shellfish has been shown to be a vehicle for the transmission of *Vibrio cholerae*. The recent cholera epidemic in Naples, Italy, took 19 lives as a result of the consumption of contaminated mussels.

Vibrio parahaemolyticus

Vibrio parahaemolyticus is a naturally occurring micro-organism, widely distributed in the marine environment including sediment, water, fish and shellfish. It is the major cause of gastroenteritis in Japan, where over 50 percent of all reported seafood-borne infection is caused by this organism[9]. More recently, *Vibrio parahaemolyticus* has been found to cause sores on swimmers[10], and diseases of crustaceans[11].

Several surveys have shown that *Vibrio parahaemolyticus* is frequently present in shellfish and growing waters in Canada and the United States[12,13]. However, the population of *Vibrio parahaemolyticus* detected from the shellfish was so low that it would be unlikely to cause any potential food poisoning. Smith[14] reported that the number of ingested *Vibrio* cells necessary to cause clinical disease appears to be about 10^6-10^9. Thus, the lack of frequent episodes of *Vibrio parahaemolyticus* infection in North America, compared with Japan, must be attributed to prevention of growth of *Vibrio parahaemolyticus* in raw or processed shellfish. Only one confirmed outbreak of *Vibrio parahaemolyticus* in North America has occurred, in Maryland, where 320 persons became ill after consuming contaminated crabs.[15].

Viruses

Enteric viruses, including infectious hepatitis, poliovirus, coxsackie viruses, ECHO viruses, adenoviruses, reoviruses and more than 100 other types of viruses, are known to be excreted in human feces[16]. The presence of enteric viruses in estuarine and ocean waters has been well documented[17,18]. However, documentation of waterborne viral epidemics has largely been limited to infectious hepatitis, mainly because of the explosive nature of these outbreaks and their characteristic symptomatology. Other waterborne virus disease outbreaks are not so easily recognized or well documented because many viruses cause inapparent or latent infections which are difficult to recognize as being waterborne. Epidemiological techniques are not sensitive enough to detect low level transmission of viral diseases through water. These reasons probably account for the fact that non-specific gastroenteritis makes up more than 50 percent of all waterborne disease outbreaks.

The average enteric virus density in domestic sewage has been estimated to be about 700 viruses per 100 ml, but as many as 46,000 per 100 ml have been detected in some untreated sewage[16]. The amount of virus present in sewage is highly variable, and depends on such factors as the incidence of disease in the community, hygienic level of the population, the socio-economic level, and time of year. Enteric viruses can survive in some numbers through sewage treatment and chlorination. They can also survive in sea water for a few days to several weeks[18].

The discharge of virus into the marine environment from sewage outfalls has always been an area of concern from a recreational standpoint, and a threat to the commercial shellfish industry. It has been well established that shellfish from polluted waters may serve as a vehicle for the transmission of hepatitis and gastroenteritis. Hundreds of cases of shellfish transmitted hepatitis and typhoid have been reported[19]. A recent outbreak of infectious hepatitis in Houston involved 263 people with the consumption of contaminated oysters from Louisiana[20].

Other Waterborne Pathogens

Other waterborne bacterial infections are less common or are not fully understood. Several of the responsible micro-organisms, such as *Pasteurella* and *Leptospira*, are normally associated with domestic and wild animal populations. These organisms gain access to the blood stream of humans through skin abrasions, or mucus membranes, to produce acute infections.

Pseudomonas aeruginosa is an opportunistic pathogen of man and animals which has been implicated in a variety of ear and skin infections among swimmers. The major source of *Pseudomonas aeruginosa* in waters appears to be the fecal wastes of man and animals[21].

Among the most common parasites found in waste water and surface waters are the amoeba, *Entamoeba histolytica* and the flagellated protozoan, *Giardia lamblia*. Amoebiasis is an intestinal infection caused by *Entamoeba histolytica*. *E. histolytica* has the ability to encyst, and these cysts subsequently enter the environment in infected feces. When the encysted amoeba re-enters a susceptible host, usually in contaminated food or water, it germinates in the gut and causes infection.

Giardia lamblia is a causative agent for giardiasis. The disease ranges from subclinical to severe maladsorption of the gut. The parasite produces cysts that are passed with feces and spread to other hosts through fecal contamination. Most documented outbreaks of giardiasis in North America have been associated with contamination of municipal water supplies and recreational waters[22].

Candida albicans is a pathogenic yeast that may be found in fecal contaminated waters. This organism is known to cause infections of the mouth, vagina and skin. Briscon[23] indicated that there was an increasing incidence of vaginal infections from *Candida albicans* in women bathing in polluted coastal waters.

SOURCES OF MICROBIAL CONTAMINATION

The main sources of microbial contamination fall under the categories of point and non-point sources.

Point sources are generally described as those which discharge to the receiving water body through a discrete pipe or ditch. Point sources of microbial contamination include discharges of untreated as well as treated municipal and industrial wastewater. Where discharges are treated and disinfection is practiced, microbial levels are generally low but not eliminated. In wastewater treatment we try to reduce the levels of pathogenic micro-organisms in the discharge by disinfection, but we do not completely eliminate the presence of micro-organisms, as would be the case in sterilization. The presence of solids and soluble organic matter in the treated wastewater increases the disinfection demand, and because of the protective nature of materials such as suspended solids, reduces the efficiency of the disinfection process.

In the Atlantic Provinces, the most recent statistics indicate that approximately 55% of our population is served by sewers which collect the wastewater, and approximately 20% of the population is served by central wastewater treatment.

About 788,000 people therefore discharge municipal wastewater to collection systems that are not connected to wastewater treatment facilities. If we were to assume an average per capita wastewater generation of 100 gpd, approximately 78,800,000 gallons of untreated wastewater are being discharged daily into our environment from communities that have collection systems. In terms of organic loading, assuming an average BOD_5 of 200 mg/L, approximately 80 tons of BOD_5 would be discharged daily.

Smaller point sources of contamination also exist from cottages and summer homes that lack treatment. These sources are scattered along our coastline and are very difficult to attach numbers to. They exert effects most notably on the commercial shellfish harvest, since they contribute to the shellfish area closures which exist.

Another point source of microbial contamination is effluent from fish and food processing plants. At the present there are approximately 500 fish plants in the Atlantic Provinces which discharge their wastes to the aquatic environment.

In recent years, it has become more and more evident that point sources of pollution are only a portion of the total pollution problem. It has been realized that the cleaning up of direct municipal and industrial discharges may not bring about dramatic improvements in water quality in some estuaries. Non-point sources of pollution account for more than 50% of the total water quality problem[24]. Non-point sources of pollution include diffuse or dispersed loadings to receiving waters from sources related to the activity of man and to natural processes. Such sources do not enter at discrete, identifiable locations and are difficult to measure or define. This type of pollution cannot be corrected by the usual abatement measures and could be a major source of microbial pollution to the Atlantic coastal shellfish growing areas[25]. The main non-point sources of microbial contamination are urban runoff, landwash from agricultural lands, and wild animals and birds. All these sources contribute to the closure of shellfish areas for the harvesting of shellfish.

Urban runoff includes surface runoff from residential, commercial and industrial surfaces, and overflows from combined sewage systems which may include bypasses from overloaded pumping stations and treatment facilities. In this case, the microbial contamination would be the same as discharging diluted, untreated, municipal wastewaters.

The effects of landwash from agricultural lands have become quite apparent in the shallow estuaries in the Atlantic Provinces. The resulting shellfish closures are caused by runoff and may thus be regulated by the intensity and duration of the rainfall which occurs. During dry weather the area may be open for shellfish harvesting, but after a given intensity and duration of rainfall the area is closed for a specified period of time, usually seven days.

Microbial contamination due to wild animals and birds may not be large relative to other sources of non-point contamination, but some shellfish closures that have high bacterial counts and no other apparent sources of contamination exist.

Boats can be considered non-point sources of microbial pollution in bays, especially during seasons when traffic is heavy and boats with people are moored overnight. Sewage discharged from vessels causes bacteriological contamination of receiving waters and has resulted in seasonal closures of shellfish harvesting areas[26].

EFFECTS OF BACTERIAL POLLUTION ON COASTAL WATERS

The main effects of sewage discharged into coastal waters are those influencing the public health and fisheries. Bathing beaches are widely distributed along the coasts of Atlantic Canada. However, because of the relatively low water temperature in the North Atlantic, the most important recreational areas are found on the Northumberland shore and Prince Edward Island. Nevertheless, during the summer months, some beaches in all areas are subject to very heavy use. The discharge of sewage into coastal areas can impair water quality on bathing beaches and constitute a potential threat to public health.

Some bathing beaches in the Maritimes are occasionally closed to swimming in the summer due to bacterial contamination. Although it is very difficult to substantiate any public health risk associated with swimming in sea water from epidemiological studies, the potential for infectivity of enteric pathogens in the sea is not in doubt[19]. Dilution of the infective agents, the small volume of water ingested by swimmers, and the large inoculum required to cause overt symptoms of clinical disease, act together to prevent a significant problem. It is possible that an epidemiological association may be missed, even when infections are noted, and subclinical infections may result because of small infective doses. The working group convened by the World Health Organization[27], to consider guides and criteria for recreational quality of beaches and coastal waters, concluded that potential health risks do exist in connection with bathing or swimming in polluted coastal waters. The guideline most commonly used in Canada to assure the safety of bathing waters is that the fecal coliform density should not exceed a log mean of 200 per 100 mL, and no more than 10 percent of the samples should exceed 400 per 100 mL.

Domestic waste discharges have a negative effect on the shellfishery in Atlantic Canada. Molluscan shellfish (oysters, clams and mussels), occurring mainly in the estuaries of the Maritimes, are highly susceptible to fecal pollution from the discharge of untreated or poorly treated municipal and private sewage disposal systems, and industrial and agricultural wastes. By virtue of their ability to concentrate and accumulate bacteria, viruses and toxic chemicals from the overlying waters, shellfish may become contaminated even at a considerable distance from the point of sewage discharge. The potential health hazard of consuming shellfish harvested from contaminated waters is particularly high, because shellfish are often eaten raw or partially cooked along with their gastrointestinal tract.

To ensure that shellfish harvested from Atlantic coastal waters are safe for human consumption, the Cooperative Canadian Shellfish Sanitation Program, administered jointly by the Environmental Protection Service and the Department of Fisheries and Oceans, requires that all shellfish growing areas be approved prior to the harvest of shellfish for human consumption. Shellfish growing areas in the Maritimes are classified as "approved", "conditionally approved" or "prohibited" on the basis of the sanitary and bacteriological water quality conditions of the area. The bacteriological criteria for an approved shellfish area requires that the median "Most Probable Numbers (MPN) of fecal coliforms in water should not exceed 14 per 100 mL, and no more than 10 percent of samples should exceed 43 per 100 mL, for a five tube decimal dilution test". Shellfish growing areas may also be closed to shellfish harvesting because of the presence of paralytic shellfish poison (PSP) produced by the dinoflagellate - *Gonyaulax excavata*. A shellfish area is closed when the shellfish toxin levels exceed 80 µg/100 g of tissue. The Bay of Fundy coast of eastern Canada is usually closed for shellfish harvesting during the summer months because of the presence of PSP.

Over the years, fecal contamination of actual and potential shellfish growing areas by municipal, industrial and agricultural waste discharges has increased with increasing land use and coastal zone development. Consequently, a fairly large proportion of Atlantic shellfish growing areas have been closed for shellfish harvesting because of water quality deterioration.

In 1981, there were 240 shellfish growing area closures in effect for the Maritimes (Figure 1), comprising a total surface area of approximately 100,000 hectares. There were 122 shellfish closures in Nova Scotia, 53 in New Brunswick, 65 in Prince Edward Island, and none in Newfoundland. Presently, there is no commercial harvesting of shellfish in Newfoundland. Unless adequate measures to control the pollution of coastal water are taken, further reductions in the acreage of available shellfish areas for harvesting will inevitably result, thereby adversely affecting local economics. Existing sewage treatment facilities do not ensure complete shellfish protection. Raw sewage generally has an average coliform density of 4.6×10^7 per 100 mL[28]. The highest possible reduction of 99 percent which could be achieved in conventional secondary sewage treatment systems would still leave 4.6×10^5 coliform per 100 mL in the effluent. Thus, it is essential that, if sewage treatment plant effluents are to be discharged into the vicinity of shellfish growing waters, disinfection of the effluent is required.

The adverse economic impact of coastal pollution on the local shellfish industry is illustrated at Caraquet Bay in northeastern New Brunswick, which has one of the most important oyster fisheries in the Maritimes. In 1972, an outbreak of gastroenteritis in Quebec was traced back to oysters from Caraquet Bay, and the area was closed to shellfish harvesting.

Subsequent pollution surveys in 1973, 1974 and 1977 identified a number of domestic and agricultural sources of pollution affecting water quality in Caraquet Bay. As a result of these surveys, approximately 6,000 acres in Caraquet Bay were closed to direct harvesting of shellfish because of the presence of fecal pollution.

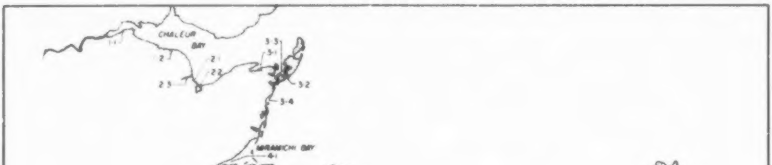
CHALEUR BAY
MIRAMICHI BAY
PRINCE EDWARD ISLAND
NEW BRUNSWICK
NORTHUMBERLAND STRAIT
GEORGE BAY
CHIGNECTO BAY
MINAS BASIN
NOVA SCOTIA
MARITIME SHELLFISHERIES
AREAS CLOSED DUE TO FECAL CONTAMINATION
AREAS CLOSED DUE TO PSP DANGER

Figure 1: Shellfish contaminated areas in the Maritimes.

The oyster resource in Caraquet Bay has been estimated at approximately $3 million (1982), while other species in the area have not received detailed assessment. It was estimated that, if the pollution problem in Caraquet Bay were eliminated, the annual production of the oyster industry in 1980 for the area could have increased from $500,000 to 2 million dollars.

REFERENCES

1. Dibble, J.T. and R. Bartha, 1979. Effect of environmental parameters on the biodegradation of oil sludge. Applied and Environmental Microbiology, 37:729-739.

2. Hellman, H., K. Klein and H. Knopp. 1966. Investigations of the suitability of emulsifiers for the cleansing of oil from water. Dtsch. gewasserk. Mitt. 11:91-95.

3. Mills, A.L. and R.R. Colwell, 1977. Microbiological effects of metal ions in Chesapeake Bay water and sediment. Bulletin of Environmental Contamination and Toxicology, 18: 99-103.

4. Blakemore, R.P., 1978. Effects of polychlorinated biphenyls on macromolecular synthesis by a heterotrophic marine bacterium. Applied and Environmental Microbiology, 35:329-336.

5. Woollen, A., 1974. Salmonellosis is an increasing world problem. Food Manufacture, *3*, October.

6. Tennant, A.D., R. Toxopeus, J.A.P. Bastren, M. Beauchamp and J. Vandwint. 1975. *Salmonella* in poultry plant effluents. EPS Technical Report, EPS-4-OR-75-1.

7. Rosenberg, M.L., K.K. Hazlet, J. Schaefer, J.G. Wells and R.C. Pruneda. 1976. Shigellosis from swimming. J.A.M.A., 236:1849-1852.

8. DuPont, H.L. and R.B. Hornick, 1973. Clinical approach to infectious diarrheas. Medicine, 52:265-270.

9. Sakazaki, R., M. Iwanami and F. Fukumi. 1963. Studies on the enteropathogenic facultatively halophilic bacteria, *Vibrio parahaemolyticus*. Japan Journal of Medical Science, 21:313-324.

10. Van Gravenitz, A. and G.O. Carrington, 1973. Halophilic *Vibrio* from extraintestinal lesions in man. Infection, 1:54-58.

11. Krantz, G.E., R.R. Colwell and E. Lovelace, 1969. *Vibrio parahaemolyticus* from the blue crab *Callinectes sapidus* in Chesapeake Bay. Science, 164:1286-1287.

12. Thompson, W.K. and C.L. Thacker, 1972. Incidence of *Vibrio parahaemolyticus* in shellfish from eight Canadian Atlantic sampling areas. J. Fish. Res. Bd. Can., 29:1633-1635.

13. Bartley, C.H. and L.W. Slanetz, 1971. Occurrence of *Vibrio parahaemolyticus* in estuarine waters and oysters of New Hampshire. Applied Microbiology, 21:965.

14. Smith, M.R., 1971. *Vibrio parahaemolyticus*, Clin. Med., 78:22.

15. Anon., 1971. *Vibrio parahaemolyticus* gastroenteritis - Maryland. Morbidity Mortality Weekly Report, 20:356.

16. Gerba, C.P., C. Wallis and J.L. Melnick, 1975. Viruses in water: the problem, some solutions. Environmental Science and Technology, 13:1122-1126.

17. Goyal, S.M. C.P. Gerba and J.L. Melnick, 1979. Human enteroviruses in oysters and their overlying waters. Applied and Environmental Microbiology, 3:572-581.

18. Metcalf, T.G. and W.C. Stiles, 1967. Survival of enteric viruses in estuary water and shellfish. *In:* G. Berg ed. - Transmission of Viruses by the Water Route. Wiley-Interscience, N.Y. p. 439-447.

19. Mosley, J.W., 1974. Epidemiological aspects of microbial standards for bathing beaches. *In:* Discharge of Sewage from Sea Outfalls. H. Gunnerson, Pergamon Press, London, p. 85-93.

20. Portnoy, B.L., P.A. Mackowiak and C.T. Caraway, 1975. Oyster associated hepatitis, failure of shellfish certification program to prevent outbreak. J.A.M.A., 223:1065-1068.

21. Geldreich, E., 1974. Microbiological criteria concepts for coastal bathing waters. Ocean Management, 3:225-248.

22. Craun, C.F., L.J. McCabe and J.M. Hughes, 1976. Water-borne disease outbreaks in the United States, 1971-1974. Jour. A.W.W.A., 68:420-424.

23. Briscon, J., 1975. Yeasts and fungi in marine environments. Societe Francaise Mycologie Medicale Bulletin, 4:159-162.

24. Novotny, V. and G. Chesters. 1981. Handbook of Non-point Pollution. Van Nostrand Reinhold, N.Y., 1981.

25. Menon, A.S. 1981. The application of NSSP criteria to non-point sources of pollution. Presented at the Technical Subcommittee Meeting of the Interdepartmental Shellfish Meeting, October 17, 1981, Ottawa, Ontario.

26. Kay, B.H. 1982. The effect of sewage discharges from anchored pleasure boats on B.C. shellfish growing areas. Environmental Protection Service, Regional Program Report, 82-10.

27. World Health Organization, 1975. Guides and criteria for recreational quality of beaches and coastal waters. Report of a Working Group. W.H.O., Copenhagen.

28. Clarke, N.A., G.Berg, T.W. Kabler, and S.L. Chang. 1962. Human enteric virus in water: source, survival and removability. Proc. 1st Int. Conf. Water Pollution Research, London, 2:523.

RESOLUTION OF MARINE POLLUTION PROBLEMS

Alan McIver*

The four keys to the control of marine pollution are understanding, education, legislation and technology. Perhaps a fifth might be added, money wisely invested!

Pollution of the oceans is generally accepted to mean, "The introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting or likely to result in such deleterious effects as harm to living resources or marine life in general, hazards to human health, hindrance to marine activities including fishing and other legitimate use of the sea, impairment of quality for use of sea water, and reduction of amenities".[1]

UNDERSTANDING

Marine pollution is recognized as a process or multiplicity of events, some minor, some of catastrophic scale, all contributing in some way to the degradation of the marine environment for a greater or lesser period of time. Of greatest concern are those pollutants with long lasting or irreversible effects. Understanding the process and mechanisms by which contaminants enter into the marine environment is critical in redressing the problems in much the same way as knowing where the leaks are when trying to get a dry basement. The methods of resolving the problems are quite similar. Usually, they require following the signs to the sources and resolving the problem at the source. Figure 1 illustrates the scope of the problem, and it is evident that there is not, nor can be, a panacea. Mitigation will result from tackling each problem with the best available scientific, technical, educational and legislative tools until an impact is made on each problem area.

Internationally, the health of the oceans has been the focus of study of the Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) formed in 1969 under the auspices of a number of United Nations' organizations.[2,3] Progress has been made in the last decade toward defining the extent of the problem, but progress has been slow, hampered by the fact that in oceanic water the contaminants present are in such small quantities that we are at the threshold of detectability in many cases, and the degree of accuracy and reliability in both sample collection and analysis is often suspect. Nevertheless, enough data has been collected with credible accuracy to give us at least a rudimentary baseline. Sampling protocols, quality control procedures for analytical laboratories, development of dependable reference samples, technological advancements in instrumentation and experience, all work toward the development of more and more reliable qualitative evaluations of the extent of contamination of the marine environment by the Pandora's boxful of substances which infiltrate that medium. There is always a danger of assuming that because the presence or effect of a contaminant is difficult to detect, it is therefore harmless. Pioneer workers with radium paid a heavy price as a result of this kind of assumption.

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The measurement and understanding of the pollution of the seas may be achieved by the sampling of sea water, or indirectly by the analysis of adjacent media, such as sediments and shore, at their marine interface. Other indirect indications of contamination are the levels of contaminants present in the biota that live in, on, beside, above or below the sea.

Point source pollution can often be detected by such analyses, but for the interpretation of most data, it is essential that baseline or reference data be gathered against which new data can be compared to permit the detection of change in space or time.

The responsibility of evaluating and protecting the marine environment is spread among various departments of the Canadian government, such as Environment Canada, Transport Canada, Energy, Mines and Resources (through the Canada Oil and Gas Lands Administration, COGLA), and Fisheries and Oceans Canada. Special funds have been earmarked for scientific studies, such as the Baseline Studies Fund, administered by Environment Canada and which, on Canada's east coast, has been focussed almost exclusively on the quality of the environmental impact of offshore oil and gas development, and particularly in establishing the baseline levels of environmental conditions which could interact with man-made structures introduced into the marine environment.

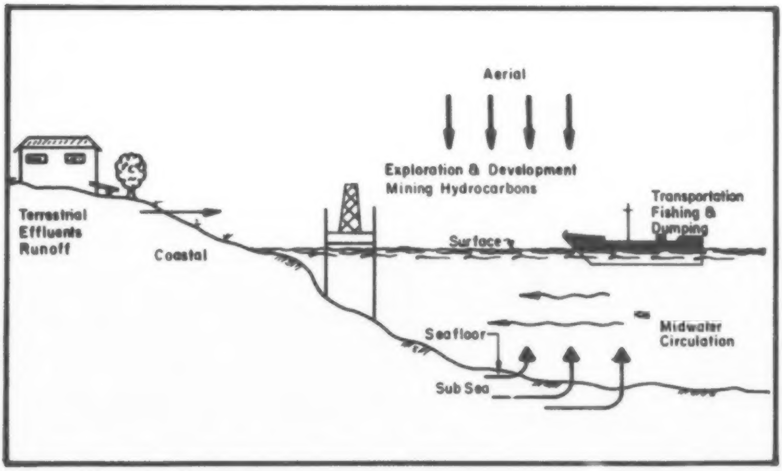

Figure 1: *Sources of ocean pollution.*

The Government of Canada has recently established the Environmental Studies Revolving Fund administered by the Canada Oil and Gas Lands Administration (COGLA) in support of oil and gas exploration and development. This 30 million dollar renewable fund (for all of Canada), although designed, in part, to determine the effect of the environment upon offshore structures, has also in its purpose the requirement to evaluate the impact of oil and gas development upon the marine environment. An essential tool in this regard is the development of means of portraying the existing environment and resources to assist in evaluating change. Examples are long term ice, weather and wave hindcasts and predictions, coastal resource inventories and sensitivity maps. Contingency plans and the use of remote sensing to detect changes in the environment are also being rapidly developed. An Energy Research and Development Fund[4] ($200 million for Canada) is another source of federal funds being used to better define the sources and extent of existing and potential marine pollution, measuring such factors as the impacts of abandoned drilling sites on the health of the sea floor.

EDUCATION

Education has played an important part in the transfer of concern from the scientific and technological communities to industrial and governmental managers and to the public at large regarding marine environmental issues. The concerns of the sixties and development of greater environmental conscience by the government in the seventies were a direct result of the increased awareness through education via the media and the printed word. "Silent Spring"[5] and "The Greening of America"[6] sounded a warning which was heard by many. This educational process has grown to the extent that now public hearings are required for major federally supported projects referred under the Federal Environmental Assessment and Review Process. This process is often combined with a similar provincial one. Public interest groups have grown in power and effect so that major projects such as marine transportation of oil or gas and offshore developments such as oil and gas in the Beaufort Sea, on the Scotian Shelf, Grand Banks, or Gulf of St. Lawrence could be constrained until public and private concerns regarding potential environmental impacts are resolved.

Developers are now, more than ever before, seeking to acquaint those potentially affected by new projects with the environmental, social and economic implications of their developments. More than ever before, they are being kept to commitments by an informed public. Government departments are committed to public involvement in policy development which can only result in a better informed public and tighter controls on environmental protection. Canadian universities also play an essential role in teaching environmental skills.

Critical in the effective prevention of pollution is the provision of training for the safe handling of products, and instruction about environmental concerns. For example, as part of an unusual program to use chemical explosives for shallow water seismic geosurveys, an instruction session for employees on the environmental concerns and implications has been included by the proponent company to explain why certain precautions must be employed.

Codes of good practice have been developed and are used by industrial organizations such as the Petroleum Association for a Clean Environment (PACE). Transport Canada and Environment Canada have also developed training programs encouraging pollution prevention.

LEGISLATION

The influence and authority of a coastal state increases as one approaches the shore.

In Canada, the protection of the marine environment is primarily a federal responsibility, first under the British North America Act, 1867, and now under the Constitution Act, 1982. The extent of Canadian waters, like the extent of any coastal nation's sovereignty, is the subject of complex negotiations and discussion through the mediation of the United Nations Organization. After eight years of negotiations, the United Nations adopted a treaty on the Law of the Sea on April 30, 1982. This agreement will come into force when it is ratified by 60 nations and will become law for all countries, including those that do not sign the treaty. The essential issues are legal rights, freedom of navigation for military and commercial vessels, environmental protection, resource access and control, and procedures governing marine science. Under the treaty, within the territorial seas, which will extend twelve nautical miles outward from a baseline measured headland to headland (Figure 2), the coastal state will have the exclusive right to govern activities. Waters within the headlands are defined as internal waters. In Canada the relative federal and provincial jurisdictions and authorities within these waters is still a subject of dispute, particularly with respect to mineral rights. This is also true for the whole Continental Shelf which, in eastern Canada, is a large area extending in some places in excess of 300 nautical miles from the coast.

Other international conventions govern the control of marine pollution in general and, therefore, influence Canada in the regulation of marine pollution in the northwest Atlantic. Canada has maintained a fairly high profile amongst nations voicing environmental concern, spurred perhaps by our desire to reinforce jurisdictional authority in the North.[4] But for whatever reason, Canada has supported the efforts of the Intergovernmental Maritime Organization (IMO) [formerly Intergovernmental Maritime Consultative Organization (IMCO)] and the United Nations Environmental Program (UNEP). Significant conventions established were the 1954 Oilpol Convention, the 1973 Marpol Convention, and the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Materials (otherwise known as the London Dumping Convention, 1975). The Marpol Convention was designed to reduce the likelihood of catastrophic and chronic oil spills and was reinforced in its formation by the devastating pollution caused by the foundering of the Torrey Canyon in 1967. Protocols were established in 1978 relating to design, construction and equipment of vessels. The Canadian government is presently revising existing Canadian legislation to meet these criteria. Responsibility for the control of pollution from vessels, a major source of marine oil pollution, lies mainly with Transport Canada[7] and is enforced under the Canada Shipping Act, Section 20 in particular. Protection is also legislated on the east coast through the Eastern Canada Traffic Regulations (ECAREG, 1978). These regulations require all vessels over 500 gross registered tons to request clearance and provide information regarding name, destination, nature of cargo, and mechanical, navigational and/or equipment deficiencies or malfunction before entering Canada's territorial sea from the high seas. Infractions can result in stiff penalties.

The passing of the Canada Oil and Gas Act (1982), and the promulgation of regulations under this Act which will follow, will provide a firm foundation for establishing and enforcing environmental protection measures and the production of contingency plans to be put in effect should a catastrophic spill occur. Operational spills and pollution from other sources such as drilling muds, use of biocides, process water, etc., are controlled by on-rig inspections by a COGLA Inspector advised by Environment Canada officials.

It should now be evident that pollution control is not vested in a single authority in Canada. However Environment Canada has a lead role to play, having an umbrella responsibility for providing coordination and advice and seeks to influence proper procedures and good housekeeping practices more by advocating than by actual enforcement. Nevertheless, enforcement mechanisms do exist and are used on occasion. Support for this role is found in Sections 31 and 33 of the Fisheries Act, which provide for the protection of fish habitat and the prohibition of disposal of deleterious substances in the sea.

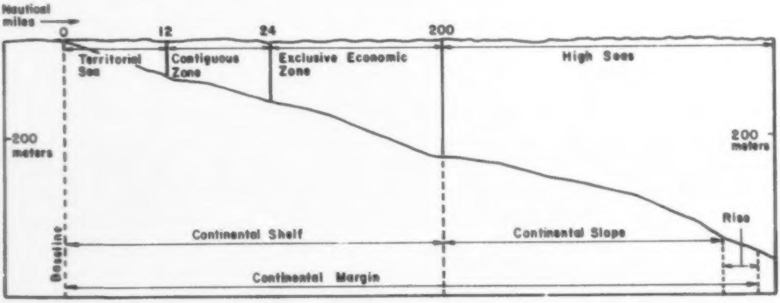

Figure 2: *Jurisdictional areas proposed by the Third United Nations Conference on the Law of the Sea*[3].

Environment Canada also enforces the Ocean Dumping Control Act (1975), which puts into Canadian law the conditions of the London Dumping Convention (1975) and provides a set of regulations by which it may be administered. Essentially the Act prohibits the disposal of contaminants such as heavy metals, persistent plastics, organohalogens, oils and radioactive material in concentrations above regulated levels, and limits disposal of a range of potentially harmful materials in the sea unless they can be rapidly rendered harmless.

On the east coast, the greatest number of permits issued have been for dredge spoil disposal. Enforcement of the Act has, however, resulted in significant protection of the marine environment in such areas as Dalhousie, New Brunswick (control of cadmium contamination); Miramichi estuary, New Brunswick (control and monitoring of a 3.5 million cubic metre dredging project to enlarge the shipping channel); and dredge spoils disposal from the Saint John Harbour, New Brunswick, an active, tidal east coast harbour in the Bay of Fundy.

Other Acts administered by Environment Canada which protect some aspects of the marine environment or marine life include the Migratory Birds Convention Act, the Canada Wildlife Act, the Canada Parks Act and, indirectly, the Contaminants and Clean Air Acts.

Input of contaminants to the marine environment is perhaps greatest from land. Here legislation is more diverse and complicated. Canada lacks a comprehensive coastal zone management scheme and, apart from the Canada Water Act (1969), also lacks the legislation which might provide a more effective framework for controlling coastal pollution, although numerous federal departments such as Fisheries and Oceans, Transport and Environment have some involvement. The respective jurisdiction of federal and provincial governments is not always clear, and varies from province to province. Generally "end of pipe effluents" are controlled by the provincial governments who must, in some instances, adhere to minimum federal standards or guidelines. These include the control of mining wastes, and pulp and paper industry effluents. Severe localized pollution has also resulted from steel plant and coal industry operations. Provincial and municipal regulation controls the disposal of sewage into the marine environment, although few if any major treatment plants exist in eastern Canada.

Although this list is not exhaustive, it underlines the existence and potential power of a legislative mechanism for controlling the pollution of the Canadian Northwestern Atlantic. The control process is cumbersome at best, and lengthy delays between understanding, education and control are common. Dr. E.D. Goldberg notes in his "Health of the Oceans" that it took two decades to halt the discharge of mercury into the coastal marine zone and to define acceptable levels in seafoods.[5] The task may be great.

TECHNOLOGY

As varied as the sources of pollution, are the mechanisms for protection or cleanup. It is generally recognized that the technology presently exists to ensure that the greater part of the environment remains unspoiled. The most effective protection of the oceans is to prevent contaminated material from being knowingly disposed of in the oceans. The establishment in the Maritimes of regulated and monitored land disposal sites is a major step in this direction. The application of science to render wastes harmless by chemical conversion is another useful tool now being employed, extending even to using acidic wastes from one process to neutralize basic wastes from another.

Oily water separators on rigs and tankers and shore treatment and disposal of bilge water and tank washings are increasingly becoming standard procedure in compliance with stringent regulations. Controlled discharge of oil beyond the 50 mile zone as proposed under the Marpol Convention, although not ideal, is considered more enforceable than a no pollution stand at present.

Other examples of technology currently employed are the use of cyclonic separators to separate solids from process plant effluents such as the stick water from fish plants, paper plants, etc.

Simple application of settling ponds prior to discharge of effluents into the sea has been one of the most effective means of cleaning effluents. This technology has been effectively applied in the Maritimes to the effluents from pulp and paper mills.

Chemical treatment of effluents such as the use of lime and flocculants is often employed to bring liquid effluents down to regulated levels, for example, the treatment of smelter discharge at Belledune, New Brunswick.

Sewage treatment has been neglected in the Atlantic Region. Marked improvement in water quality would result if secondary and tertiary processing were to be employed. New sewerage is being constructed in such major cities as Halifax and Saint John with an ultimate aim of closing the multitude of shoreline outlets and diverting them through a single major sewer. This type of modification in retrospect is very costly.

Ocean disposal can result in a minimum impact if clean material is disposed of in a carefully selected site, and if contaminated material is covered with clean material or in a manner which rapidly renders it harmless. Use of satellite aided positioning and increased knowledge make this a possibility that was beyond existing capability until recently.

COST

The question of who should pay for environmental protection and correcting the detrimental effects of pollution is one which demands an answer. The cry of the 70's was, "let the polluter pay". This was appropriate for situations where it was apparent who the polluter was, but there are many instances where environmental insult has occurred over many years, even centuries. Wittingly or unwittingly, public consent was given, or at least a blind eye was turned to the consequences of some industrial development, so that employment opportunities for communities were not lost to other areas. Sparce populations and large tracts of land or water bodies seemed at one time to have an almost limitless capacity to contain or absorb wastes and by-products. We now know this not to be so, but in many cases the environmental "sins of the fathers are visited upon the children" perhaps for many generations. The synergistic or cumulative effect of many minor pollution sources (such as domestic sewage) can add up to a significant input again not traceable to a single source. Hence the "polluter pay" principle is not always applicable, and common users must be charged with the cost through direct taxes or levies or indirectly through government intervention, again at public cost.

In the Atlantic Provinces, the environmental costs of offshore oil and gas development are shared by the proponents, government, and the general public. Industry has increasingly borne a share of the cost of determining the extent and influences of marine pollution, and as the climate of general concern has increased, has increasingly incorporated into the design and pursuit of its operations the technology and costs of protecting the environment from detrimental side effects.

As noted earlier, each major project involving significant public funding or support is required to submit to a Federal or Federal/Provincial Environment and Assessment Review Process. The proponent company and the government department which initiates the proposed development must submit an "Environmental Impact Statement" or lesser document produced at their expense as a result of a literature review and project specific research and investigation. This documents their evaluation of the expected impact of the development and proposes mitigating measures. The document is then presented to a Federal or Federal/Provincial Review Panel and is reviewed extensively by the government and private citizenry. The cost of this review is borne directly or indirectly by the public. In a sense, because the public will also be beneficiaries of the development along with the proponent company, the principle of "polluter pay" still holds.

Costs to companies required to incorporate pollution prevention devices in new or existing plants and to comply with increasingly stringent regulations will also be reflected in the final cost to the consumer, foreign or domestic.

The cost of not protecting the marine environment must also be considered. Already numerous productive clam flats are closed to harvesting due to bacterial contamination from sewage outfalls. Oil spills account for other immediately perceivable losses to the economy. Less measurable in terms of cost are the degradation of the environment aesthetics, social disbenefits and lowering of the quality of life in polluted areas. More sinister still are the possible sublethal effects, which, although not killing an organism outright, may result in stress or loss of fertility, thus although no abnormal mortalities may be observed yet significant loss of stock may result from poor recruitment of future generations. No organisms are immune to such environmental pressures.

The short term solution to marine pollution lies in developing better and more effective cleanup technology, but the long term and only truly effective solution is to eliminate pollution at the source by the development of environmentally sound methods of handling the products of our society, especially the wastes. This will require radical revision in our philosophy regarding the stewardship of our natural resources and will have to penetrate the values and lifestyle of our culture. But we have started, and with the tools of understanding, education and legislation and technology we must succeed.

REFERENCES

1. Pravdic, Velimir. GESAMP. The First Dozen Years. UNEP, 1981.

2. GESAMP (IMCO/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP JointGroupof Experts on the Scientific Aspects of Marine Pollution). The Health of the Oceans. UNEP Regional Seas Reports and Studies, No. 16, UNEP, 1982.

3. Ross, A David. Law of the Sea: How will it affect marine research? Sea Grant Today, p 5-7, Vol. 12, No. 2, March/April, 1982.

4. Energy R&D Fund Brochure. 1982.

5. Carson, Rachel, 1962. The Silent Spring. Houghton-Mifflin, Boston.

6. Reich, Charles, 1970. Greening of America. Random House, N.Y.

7. Working Group of the Interdepartmental Committee on Prevention of Pollution from Shipping in Waters under Canadian Jurisdiction. Transport Canada, Coast Guard, Ottawa, Printed June, 1981, 252 p.

8. McDorman, Ted. L. The Protection of the Marine Environment: The Canadian Record, 1967-1982. Dalhousie Ocean Studies Program, Dalhousie, Halifax, December, 1981, 35 p.

9. Goldberg, Edward D. The Health of the Oceans, UNESCO Press, 1976, 171 p.

SUMMARY AND CONCLUSIONS

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It should be clear from the foregoing chapters that the offshore Northwest Atlantic is relatively uncontaminated, as far as we can tell from comparisons with other, more obviously polluted areas such as the eastern North Sea, the Baltic or the Mediterranean. This is so for several reasons.

(1) Population density along the eastern Canadian coast is relatively low, and so sources of man-made contaminants are limited;

(2) There are relatively few industrial sources of pollution along the Canadian Atlantic coast;

(3) The Northwest Atlantic is a very dynamic environment, and any pollutants which are introduced to it tend to be dispersed rapidly and widely.

Two related points are worth emphasizing:

(1) Although offshore contamination is low, there are examples of local inshore pollution, usually associated with local industrial discharges;

(2) Much of the pollution which is found in the offshore Northwest Atlantic seems to arise from diffuse or non-point sources, such as by atmospheric transport from the Northeastern U.S.

In most instances, levels of contaminants in the offshore are not high enough to cause any obvious or acute toxic effects, but these effects may sometimes be seen in a few cases of localised inshore pollution. Levels of contaminants in fish products from the Northwest Atlantic are usually well within ranges acceptable for public health reasons, although in some instances, inshore fisheries may be closed because of unacceptably high contaminant levels.

Long term trends are difficult to distinguish. Unless there is some dramatic industrialization in the Atlantic Provinces, it seems unlikely that the local contribution to either inshore or offshore pollution will change much. Since much of the offshore pollution appears to come from distant sources, future trends in offshore contaminant levels will depend very much on economic and political developments outside the Canadian Atlantic region - especially in the industrialised Northeastern U.S. The public awareness of environmental matters which has grown throughout the 1970s may lead to tighter restrictions on industrial effluent control, and this may have some eventual impact on marine environmental quality in the region. We can already detect a response to such regulations, insofar as both organochlorine insecticide and certain petroleum hydrocarbon levels appear to be declining in the Northwest Atlantic. Long may this trend continue.

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